US Army Corps
of Engineers
Nashville District

DALE HOLLOW LAKE
CE-QUAL-W2
WATER QUALITY MODEL

FINAL REPORT
September 7, 2001

Dale Hollow Lake
CE-QUAL-W2
WATER QUALITY MODEL

Prepared for

US Army Corps of Engineers
Nashville District

Under Contract
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Prepared by

FTN Associates, Ltd.
3 Innwood Circle, Suite 220
Little Rock, AR 72211

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Philip Massirer, Dennis E. Ford

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FTN Associates, Ltd.
3 Innwood Circle, Suite 220
Little Rock, AR 72211

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13. ABSTRACT *(Maximum 200 words)*

The objective of this study was to provide a calibrated temperature and water quality model for Dale Hollow Lake suitable for addressing management related issues. CE-QUAL-W2 (Version 2.05), a two-dimensional, longitudinal and vertical hydrodynamic and water quality model, was chosen as the model for this application. Modifications to the model included the use of three algal compartments instead of one, addition of silica as a water quality constituent affecting algal productivity, and addition of an algorithm to create an output file to be used by the Animation and Graphics Portfolio Manager (AGPM) pre- and post-processor.

The model was calibrated to three years (1973, 1988 & 1991) representing dry, wet and average hydrologic conditions. The study included the development of boundary conditions for the major inflows through regression analyses.

The calibrated model was used to evaluate the impacts of point source discharges, including houseboats, on water quality conditions on Dale Hollow Lake.

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1.0 INTRODUCTION

Dale Hollow Lake water quality modeling was performed for the Nashville District, US Army Corps of Engineers (District) under Contract Number DACW62-98-D-002, Delivery Order No. 0012. The objective of this modeling effort is to provide the District with a calibrated CE-QUAL-W2 model of Dale Hollow Lake that is suitable for evaluating existing water quality conditions and temporal trends, and predicting water quality conditions in the reservoir under various management scenarios.

The CE-QUAL-W2 water quality model was selected for use because of its applicability in addressing concerns related to the District projects. The modified version of CE-QUAL-W2 developed for the District for application to Center Hill Lake was also used in this application (FTN 2001). This report describes model calibration, confirmation, and simulation of management scenarios. Results of these activities and the dominant physical, chemical, and biological processes affecting Dale Hollow Lake water quality are specifically highlighted in this report.

The report sections are arranged in chronological order with respect to steps initiated in the modeling procedure. It should be noted that steps involved in the modeling procedure are interdependent and may be repeated as part of an iterative process.

This report is organized as follows:

- Section 2.0 describes the general characteristics of Dale Hollow Lake and its watershed.
- Section 3.0 describes the development of the reservoir bathymetry file, including branches, tributaries, and outlet configuration.
- Section 4.0 describes procedures for the calibration of the water budget, hydrodynamics, temperature and water quality.
- Section 5.0 describes the different scenario simulations.
- Section 6.0 provides a list of references.

2.0 BACKGROUND

2.1 Reservoir and Watershed System

Dale Hollow Lake is a US Army Corps of Engineers project located in north central Tennessee and south central Kentucky. The multi-purpose project was authorized for construction on the basis of benefits attributed to flood control and hydropower generation (Figure 2.1). The reservoir is the result of impounding the Obey River 7.3 river miles above its confluence with the Cumberland River at Celina, TN. The reservoir became functional for flood control in October 1943. The three hydropower units were placed in service in December 1948, January 1949, and November 1953. Morphometric characteristics of the reservoir are summarized in Table 2.1. In addition to flood control and power generation, additional operating purposes include recreation, fish and wildlife, water quality, and water supply. No water use contracts exist at this time.

The Obey River drainage area above Dale Hollow dam is 935 square miles. Kentucky includes 17% of the watershed and the remaining 83% of the drainage area lies within the State of Tennessee. Mayland Lake, located in the extreme southern portion of the watershed is the only known water control structure upstream of Dale Hollow Lake. Given the distance between Mayland Lake and Dale Hollow Lake no impact to flow or water quality in the East Fork Obey River is expected.

In the "Water Quality Conditions in Dale Hollow Reservoir" report dated July 1976; the watershed is reported to be 60% forested. The urban areas of the watershed include Jamestown and Byrdstown, TN, and Albany, KY (Figure 2.1). Each of these urban areas discharge treated wastewater to Dale Hollow Lake tributaries. As reported in 1976, approximately 704 acres around the lake is in use as boat docks and recreation areas. This type of intensive development can create localized water quality problems.

In addition to the recreational areas, there is a relatively high concentration of houseboats on Dale Hollow Lake. According to lake managers, during the 1999 summer season there were 500 to 600 privately owned houseboats on the lake, and an additional 200 or more available for

Table 2.1. Morphometric characteristics of Dale Hollow Lake.

	English Units		Metric Units	
Volume, V				
Conservation Pool (631 ft msl)	857,000	ac-ft	1,056,681,000	m^3
Power Pool (651 ft msl)	1,353,000	ac-ft	1,668,249,000	m^3
Flood Control Pool (663 ft msl)	1,706,000	ac-ft	2,103,498,000	m^3
Length, L				
Flood Control Pool	61	mi	98	km
Shoreline Length, L_{SH}				
Flood Control Pool	620	mi	998	km
Surface area, SA				
Conservation Pool	21,880	ac	88	km^2
Power Pool	27,700	ac	112	km^2
Flood Control Pool	30,990	ac	125	km^2
Mean width, W ($^{SA}/_{L}$)				
Conservation Pool	0.56	mi	0.90	km
Power Pool	0.71	mi	1.14	km
Flood Control Pool	0.79	mi	1.22	km
Maximum depth, Zm (invert = 508 ft msl)				
Conservation Pool	123	ft	37	m
Power Pool	143	ft	44	m
Flood Control Pool	155	ft	47	m
Mean depth, Z ($^{V}/_{SA}$)				
Conservation Pool	39	ft	12	m
Power Pool	49	ft	15	m
Flood Control Pool	55	ft	17	m
Hydraulic residence time	343	days	343	days
Watershed area	935	mi^2	2,422	km^2
Normal pool elevation range	631-651	ft, msl	192-198	m, msl
Average annual inflow, Q	1,580	cfs	45	m^3/sec
Range of annual inflow	890-2,420	cfs	25-68	m^3/sec
Spillway elevation (crest)	651	ft, msl	198	m, msl

Notes: Average inflow and range of inflow are based on April 1919 – December 1951 Source Anon. "Draft Environmental Statement of Operation and Maintenance of Dale Hollow Lake, Obey River, Tennessee," US Army Engineer District, Nashville, TN, October 1975.

rent. As expected, most houseboats are seen on the lake during the weekends, with the greatest number, as many as 1,000, present over the Labor Day holiday weekend. The resource managers on Dale Hollow Lake attempt to enforce "no dumping" based on Title 36 of the Code of Federal Regulations, to restrict the direct discharge of untreated sanitary wastewater into the lake by

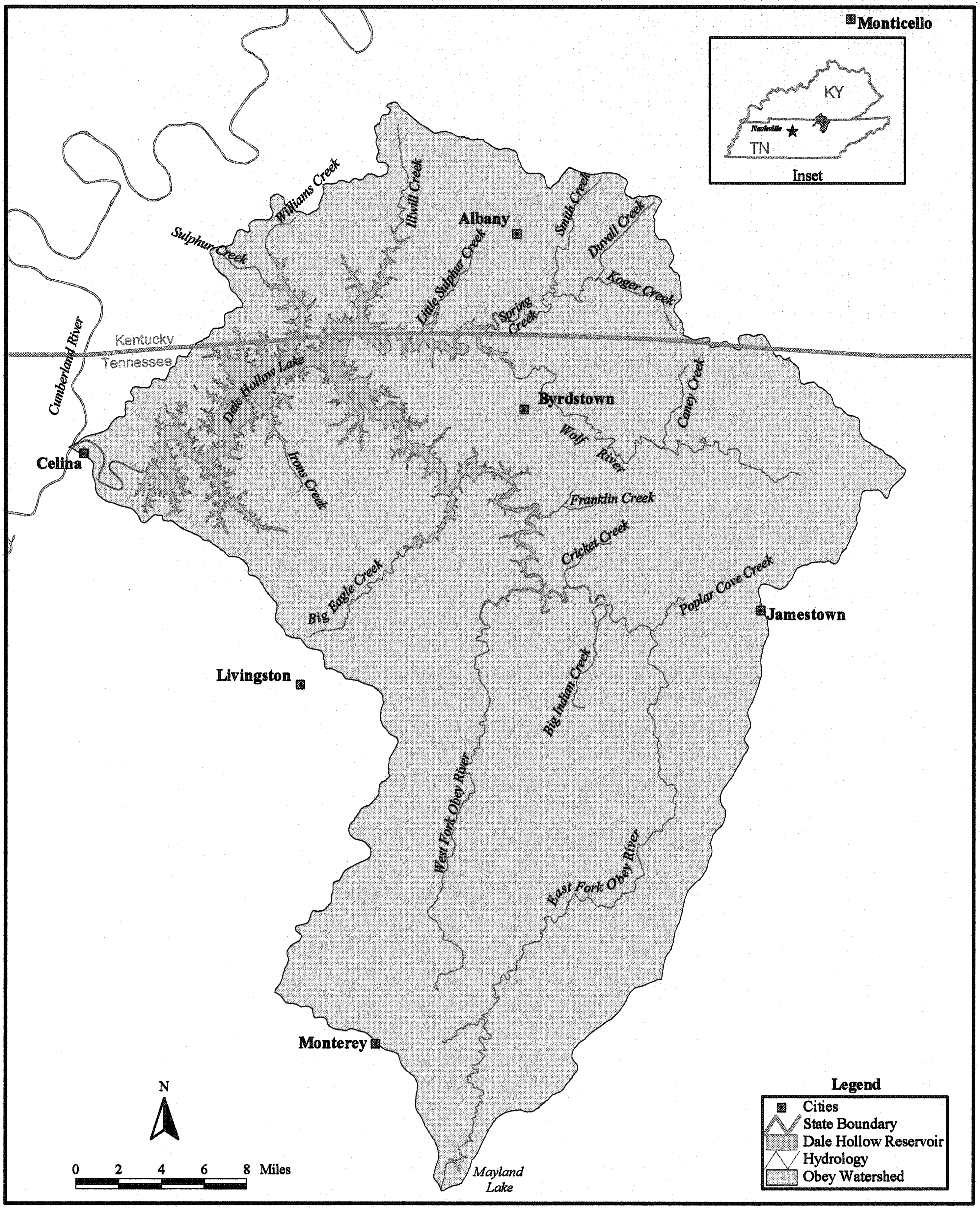

Figure 2.1 Site Location & Watershed Map Dale Hollow Lake

houseboats (pc. Jeff Hancock, US Army Corps of Engineers, September 2001). Compliance is encouraged by resource managers and marina associations. Dumping stations are provided at all marinas, however due to difficulties in policing the direct discharge 100% compliance is not realized. The number of houseboats has increased significantly in the past 10 years, and this high concentration of use and its associated impacts can greatly impact the water quality of Dale Hollow Lake. There are also a number of businesses that discharge wastewater in the watershed. Although these discharges are not shown on the map, a list of NPDES permitted discharges in the watershed is included as Appendix A. An additional water quality concern has been centered around oil and gas drilling. Lake managers have indicated an increase in interest and activity in drilling as gas and oil prices rise. In the March 1986 "Water Quality Assessment of Dale Hollow Lake and its Inflows", the East Fork Obey River inflow was singled out as a being strongly impacted by acid mine drainage. However, Dale Hollow Lake is not affected due to 6 miles of subterranean drainage between mile 26.4 and 20 of the East Fork Obey River. All mines on the East Fork Obey River watershed are now closed.

2.2 Reservoir Water Quality

Dale Hollow Lake is monomictic with summer temperature stratification and complete mixing during the winter period. Thermal stratification is present between April and November. It is during or after October that the onset of fall overturn occurs. The water column is completely mixed after the fall overturn until the start of spring stratification.

The District monitors water quality in the reservoir at a series of stations in the lake and at 9 reservoir tributaries. The locations of the stations are shown on Figure 2.2 and described in the accompanying Table 2.2. Samples are typically analyzed for nutrients and chlorophyll *a*, among other parameters.

In 1975-1976 and 1985-1986, the EPA and the District performed water quality studies on Dale Hollow Lake. In all of the studies, the objective was to assess the nutrient content and determine the water quality of the lake; i.e., it's trophic state. The District published "Water Quality Conditions in Dale Hollow Lake" in July 1976. The conclusions of the report indicate the trophic status of the lake to be oligotrophic and of good water quality. The lake is phosphorus limited with respect to productivity. The study by the EPA, "The National Eutrophication Survey", was published in June 1977. The EPA study concluded that Dale Hollow Lake was oligo-mesotrophic, which means the supply of nutrients as well as the productivity in the lake was low to moderate. Dale Hollow Lake ranked first in the overall trophic quality when compared with four other Kentucky reservoirs sampled during 1973. The trophic quality was based on a combination of six parameters, which included total and dissolved phosphorus, inorganic nitrogen, mean secchi transparency, chlorophyll *a*, and DO. In March 1986 the District published "Water Quality Assessment Dale Hollow Lake and Its Inflows". The study that served the basis of this report was performed during the period May through November 1985. The conclusions from this report continue to support the good water quality status of Dale Hollow Lake.

During 1988, the Kentucky Department of Natural Resources and Environmental Protection (KYDNR) measured water quality data on several arms of Dale Hollow Lake located in Kentucky. This data has been incorporated into the 1988 study year. The station locations are indicated in Figure 2.2 and defined on Table 2.2.

In the State of Tennessee, the only water quality station associated with the Obey River was downstream of the Dale Hollow dam and was not included in this study.

Table 2.2. Dale Hollow Lake water quality station descriptions.

Station	River Mile	Stream
Corps of Engineers		
3DAL10001	6.8	Dam, Obey River
3DAL10011	22.7	Wolf River
3DAL10012	2.7	Spring Creek
3DAL10013	6.2	Spring Creek
3DAL10014	12.6	East Fork Obey River
3DAL10015	7.5	West Fork Obey River
3DAL10016	9.5	Illwill Creek
3DAL10019	0.6	Franklin Creek[1]
3DAL10022	0.4	Big Indian Creek[1]
3DAL10023	25.7	East Fork Hamilton Creek[1]
3DAL10026	2.5	Williams Creek
3DAL10027	4.4	Irons Creek
3DAL10028	5.3	Big Eagle Creek
3DAL10029	4.2	Little Sulphur Creek[1]
3DAL10031	6.2	Sulphur Creek
3DAL20002	7.8	Dale Hollow Lake / Obey River
3DAL20003	16.7	Dale Hollow Lake / Obey River
3DAL20004	27.7	Dale Hollow Lake / Obey River
3DAL20005	32.7	Dale Hollow Lake / Obey River
3DAL20006	41.6	Dale Hollow Lake / Obey River
3DAL20007	55.8	Dale Hollow Lake / Obey River
3DAL20008	8.7	Dale Hollow Lake / Wolf River
3DAL20009	3.0	Dale Hollow Lake / Obey River
3DAL20010	1.8	Dale Hollow Lake / West Fork Obey River
USEPA		
2102A1	2.7	Spring Creek
2102B1	4.7	Little Sulphur Creek
2102C1	9.5	Illwill Creek
2102D1	20.7	Wolf River
2102E1	11.1	East Fork Obey River
2102F1	0.5	Big Indian Creek[1]
2102G1	1.0	Poplar Cove Creek[1]
2102H1	7.6	West Fork Obey River
2102J1	5.3	Big Eagle Creek
2102K1	6.8	Dam, Obey River[1]
210201	7.8	Dale Hollow Lake / Obey River
210202	17.5	Dale Hollow Lake / Obey River
210203	0.0	Dale Hollow Lake / Wolf River
210204	9.7	Dale Hollow Lake / Wolf River
210205	42.2	Dale Hollow Lake / Obey River
210206	0.0	Dale Hollow Lake / West Fork Obey River

Station	River Mile	Stream
USGS		
03414500	12.6	Obey River, East Fork
03415000	7.7	West Fork Obey River
03416000	26.2	Wolf River
KYDNR		
CLN114	5.6	Sulphur Creek
CLN115	5.6	Williams Creek / Sulphur Creek
CLN116	1.0	Illwill Creek
CLN117	N/A	Fanny's Branch of Illwill Creek
CLN118	1.0	Little Sulphur Creek[1]
CLN119	1.0	Spring Creek

[1] Station data not used in Dale Hollow water quality study.

Table 2.3 lists characteristic values of several indicator parameters that are associated with the three trophic states (modified after Wetzel 1983). The water quality data collected on Dale Hollow Lake were analyzed to develop historical ranges of the indicator parameters (Table2.3). Comparing the Dale Hollow Lake ranges with those for the various trophic states results in an oligo-mesotrophic lake classification for the reservoir.

Table 2.3. Trophic status indicators and Dale Hollow Lake status.

	Trophic State Indicator Mean Value and Range			
Indicator	Oligotrophic[1]	Mesotrophic[1]	Eutrophic[1]	Dale Hollow Lake (1971 – 1999)[2]
Summer Oxygen Distribution (mg/L)	Orthograde, Hypo. DO>2	Hypo. DO>0 to Anoxic	Clinograde, Anoxic	Hypo. DO>0 to Anoxic
Total Phosphorus (μg/L)	8 3–18	27 11–96	84 16–386	34 10-440
Total Nitrogen (μg/L)	660 300–1,630	750 360–1,390	1875 390–6,100	605 100-1,500
Secchi Depth Transparency (m)	10 5.5–28.5	4 1.5–8	2.5 1–7	3.1 0.2-6.7
Average Summer Chlorophyll *a* (μg/L)	1.5 0.3–4.5	5 3–10	15 3–75	2 0-10

[1] Modified after Wetzel (1983)
[2] Period of record varies with parameter

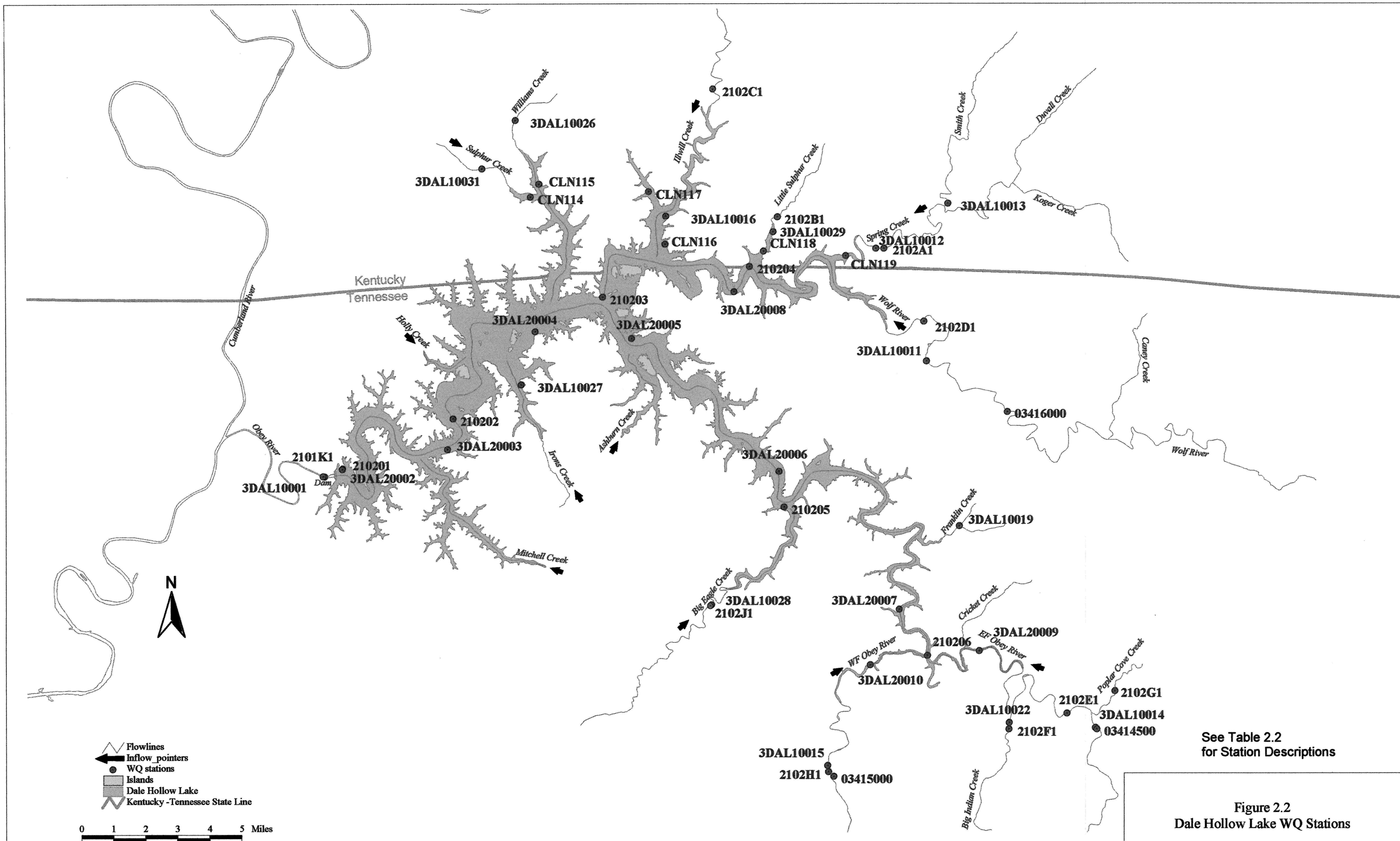

Figure 2.2
Dale Hollow Lake WQ Stations

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In many recent water quality studies, the magnitude of point and nonpoint source loadings have been found to change with time. For Dale Hollow Lake, it has been suspected that the nonpoint loadings may have increased due to changes in the land use and the increased number of houseboats in the lake. Improvements in treatment at existing point source discharges may have resulted in lowering the point source loadings.

2.3 Land Use Effect

Over the past 25 years the District has noticed a decline in the hypolimnetic dissolved oxygen (DO) conditions of Dale Hollow Lake. A review of the available land use information was done to determine if the observed decline could be related to a historical trend in the watershed of increased urban development pressure.

Land use data for the Dale Hollow watershed representing 1975 was obtained via the US EPA Basins database (USEPA 1996). The database is a compilation of data spanning the 1970's and is intended to best represent 1975. More recent land use information, 1992, was extracted from the National Land Cover Data developed from data acquired by the Multi-resolution Land Characterization (MRLC) Consortium (NLCD 2000). The MRLC Consortium is a partnership of federal agencies that produce or use land cover data. Partners include but are not limited to the USGS, USEPA, US Forest Service, NOAA, and Kentucky and Tennessee State resource agencies. A summary of the land use distribution in the watershed compiled from these two databases is included in Table 2.4.

Table 2.4. Land use changes in Dale Hollow Lake Watershed.

LAND USE	1975 % of Total Area	1992 % of Total Area	Difference
Commercial/Industrial/Transportation	0.3	0.2	-0.1
Residential/Urban	3.2	1.0	-2.2
Forested	66.3	79.4	+13.1
Agriculture	25.9	15.0	-10.8
Wetland	0.0	0.2	+0.2
Open Water	3.8	3.9	+0.1
Transitional	0.3	0.2	-0.1
Mines/Quarries/Gravel Pits	0.2	0.1	-0.1

Source Data: 1975 USEPA 1996
1992 NLCD 2000

In each data set there were similar land use classifications that could be combined to produce comparable categories. In an attempt to validate the trends indicated in Table 2.4 alternative data were obtained from the US Census Bureau (2000) and the USDA National Agricultural Statistics Service (NASS 1999). It is important to note that the NASS data is collected by county and that the 7 counties that include portions of the Dale Hollow Lake watershed comprise an area of 2,293 square miles. The Dale Hollow Lake watershed is 935 square miles. The 7 counties include the Clinton, Cumberland, and Wayne Counties in Kentucky and Clay, Fentress, Overton, and Pickett Counties in Tennessee. According to the US Census, the growth in the 7 counties making up the bulk of the Dale Hollow watershed decreased approximately 0.5% from 1980 to 1992. The land use comparison table indicates that from 1975 to 1992 the percentage of the watershed classified as urban/residential decreased 2.2%. This percent change is probably more indicative of a measurement variability and basically represents little change in urbanization, which is supported by the US Census Bureau data. As reported in Table 2.4, between 1975 and 1992 the percentage of the watershed classified as agricultural use decreased 10%. The NASS data for 1987 to 1992 , averaged over the 7 counties in the watershed, reports the amount of acreage classified as agricultural declining approximately 6%. In 1992, according to the NLCD land use classification, 15% of the Dale Hollow Lake watershed is in agricultural use. The NASS reports that in 1992 43% of the 7 counties making up the watershed is devoted to agriculture. Agricultural land is often more prevalent in upland areas. The upland area that is beyond the watershed divide, but located within the counties that include the lake, most likely accounts for this additional acreage designated as agricultural land.

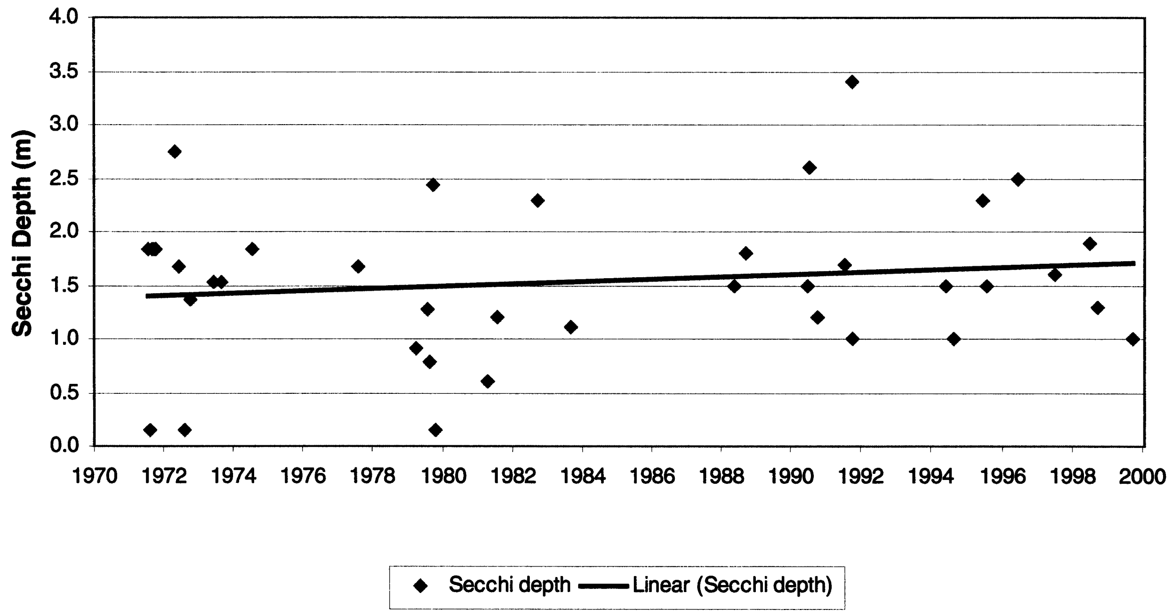
Because of the District's expressed concern about the decline of the DO in the hypolimnion, the agricultural statistics were reviewed further to see if a trend existed that more livestock and poultry were present in the watershed and could be a contributing factor. The NASS data indicated from 1987 to 1997 the number of livestock and poultry increased in 6 of the 7 countries in the range of 4% to 50%, although the acreage dedicated to livestock and poultry remained relatively constant.

For the period 1990-2000, the US Census Bureau indicates the 7 counties making up the Dale Hollow watershed area experienced an overall average population increase of approximately 10%. This report data implies that the increased urbanization in the watershed is concentrated in the years after 1992 and is therefore not accounted for in available land use databases.

In addition to the observed declines in hypolimnetic DO discussed above, the District also noticed that water clarity in the upper reaches of the Wolf River and East Fork Obey River embayments seem to be diminishing. Using Secchi Disk depth measurements at District water quality stations located in these two embayments, the data were plotted to identify historical trends. Data from both stations exhibited significant variability from year to year. Data from the Wolf River (Station 3DAL20008) appears to confirm the decreasing trend in water clarity. In contrast, data from the East Fork Obey River (Station 3DAL20009) can support either an increasing or decreasing trend in water clarity depending on whether two data points are used (6.71m and 6.1m in 1973 and 1979, respectively). If these data are considered outliners, then it would appear that water clarity is increasing in the East Fork Obey River (Figure 2.3).

In summary, the available land use classification schemes for Dale Hollow Lake do not display a trend indicative of water quality degradation. It was found that the land use classification scheme did not always agree with alternative data sources. Therefore, the assumption that changing land use is a cause of the hypolimetic DO decline could not be substantiated. A trend in reduced water clarity in upper reaches of the reservoir, may or may not be evident, depending on how the data is perceived.

East Fork Obey River



Wolf River

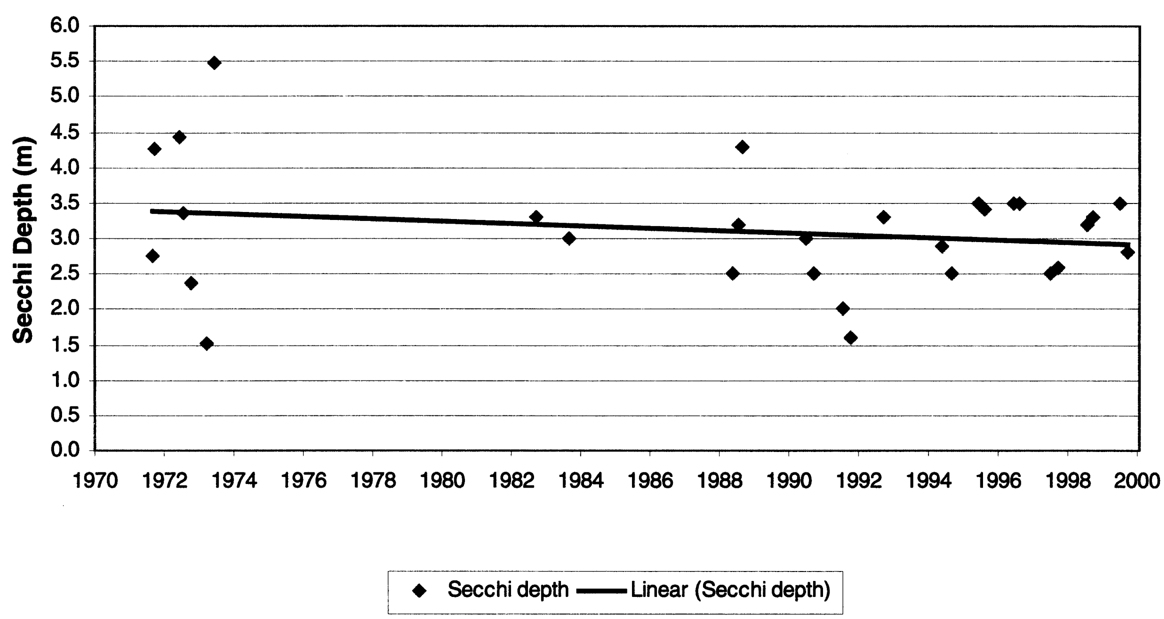


Figure 2.3. Water Clarity in the Upper Reaches of Dale Hollow Lake.

2.4 Model

A model version of CE-QUAL-W2 version 2.05 was used for this study. Modifications to CE-QUAL-W2 version 2.05 included use of three algal compartments instead of one, addition of silica as a water quality constituent affecting algal productivity, and addition of an algorithm to create an output file to be used by the Animation and Graphics Portfolio Manager (AGPM) pre- and post-processor (Loginetics Inc. 1998). The multiple algal compartments and silica algorithms were based on algorithms developed by Tom Cole and used in later versions of CE-QUAL-W2. The AGPM output algorithm was provided by Loginetics Inc. Appendix B includes printouts of these algorithms. Modifications to the model resulted in changes in the control file and in the constituent input files. Appendix B also includes a printout of the control file with the modified sections highlighted. The revised constituent order for input files is noted in Table 2.5. The constituents incorporated into CE-QUAL-W2 and their interactions are shown on Figure 2.4. Not all of these constituents were simulated for Dale Hollow Lake (see Section 4.4).

Table 2.5. CE-QUAL-W2 input order for water quality constituents.

1.	Tracer	13.	Sediment
2.	Suspended Solids	14.	Inorganic Carbon
3.	Coliform	15.	Alkalinity
4.	Dissolved Solids	16.	pH
5.	Labile DOM	17.	Carbon Dioxide
6.	Refractory DOM	18.	Bicarbonate
7.	Silica	19.	Age of Water
8.	Detritus	20.	Iron
9.	Phosphorus	21.	CBOD
10.	Ammonia	22.	Diatoms
11.	Nitrate-Nitrite	23.	Greens
12.	Dissolved Oxygen	24.	Cyanobacteria

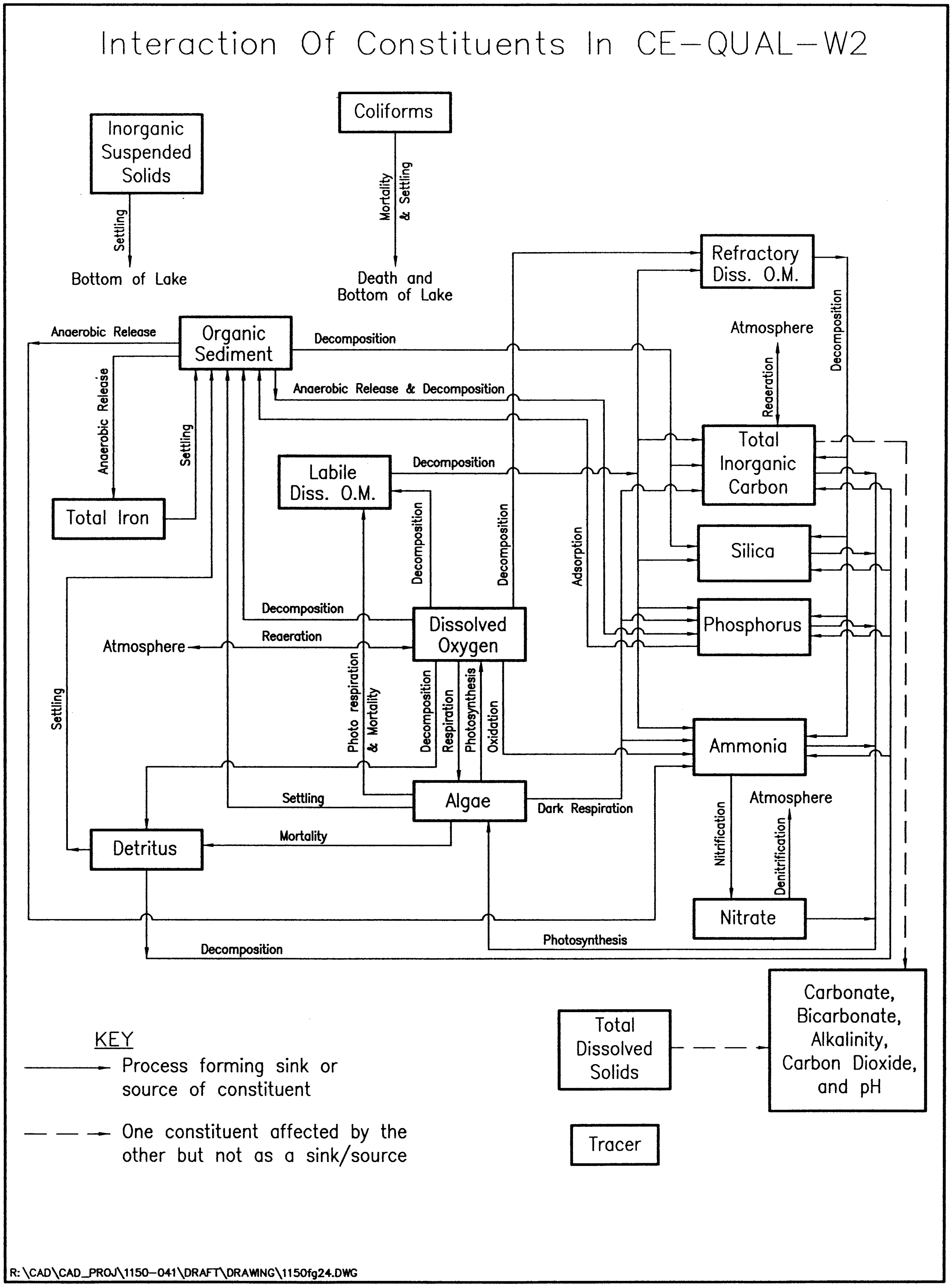


Figure 2.4. Interaction of constituents in CE-QUAL-W2.

3.0 RESERVOIR CONFIGURATION AND BATHYMETRY

A bathymetry plan was developed to describe how the reservoir would be segmented for modeling. It was submitted to the District for review and is included in this report as Appendix C. During the model setup process, several details of the bathymetry plan were modified. The development of the reservoir configuration and each part of the bathymetry input file is discussed below. A printout of the CE-QUAL-W2 bathymetry input file is included in Appendix D.

3.1 Segments and Layers

CE-QUAL-W2 conceptually represents a water body as a two-dimensional array of cells with each cell extending across the width of the water body. This "grid" of cells is determined by the longitudinal segment lengths (Δx) and layer thickness (Δz) specified by the user. The grid for Dale Hollow Lake used segment lengths (Δx) of 0.5 to 5.0 miles and a layer thickness (Δz) of 1.0 m (3.3 ft). Most of the segments are between 1.0 and 2.7 miles long. Nearly all of the segment boundaries were established at locations where cross sections were measured by the District; this was done to minimize interpolation between cross sections. A plan view of the main branch segment is shown in Figure 3.1. Layers in the grid extend from elevation 506.3 ft to elevation 663.8 ft, msl (just above the top of the flood pool). A generalized side view of the branch grids is shown in Figure 3.2. This grid is detailed enough to allow the simulation of both vertical (i.e., stratification) and longitudinal gradients in temperature and water quality constituents. Increasing the detail of this grid (i.e., making the segments shorter and/or the layers thinner) would only increase the model run time and output file sizes and it would increase the possibility of numerical instabilities at branch intersections.

Figure 3.1 also illustrates the model flow lines and conveyance channel through Dale Hollow Lake. The flow line locations were chosen based on previous studies and professional judgement. The main branch flow line indicates how water is expected to follow the main channel.

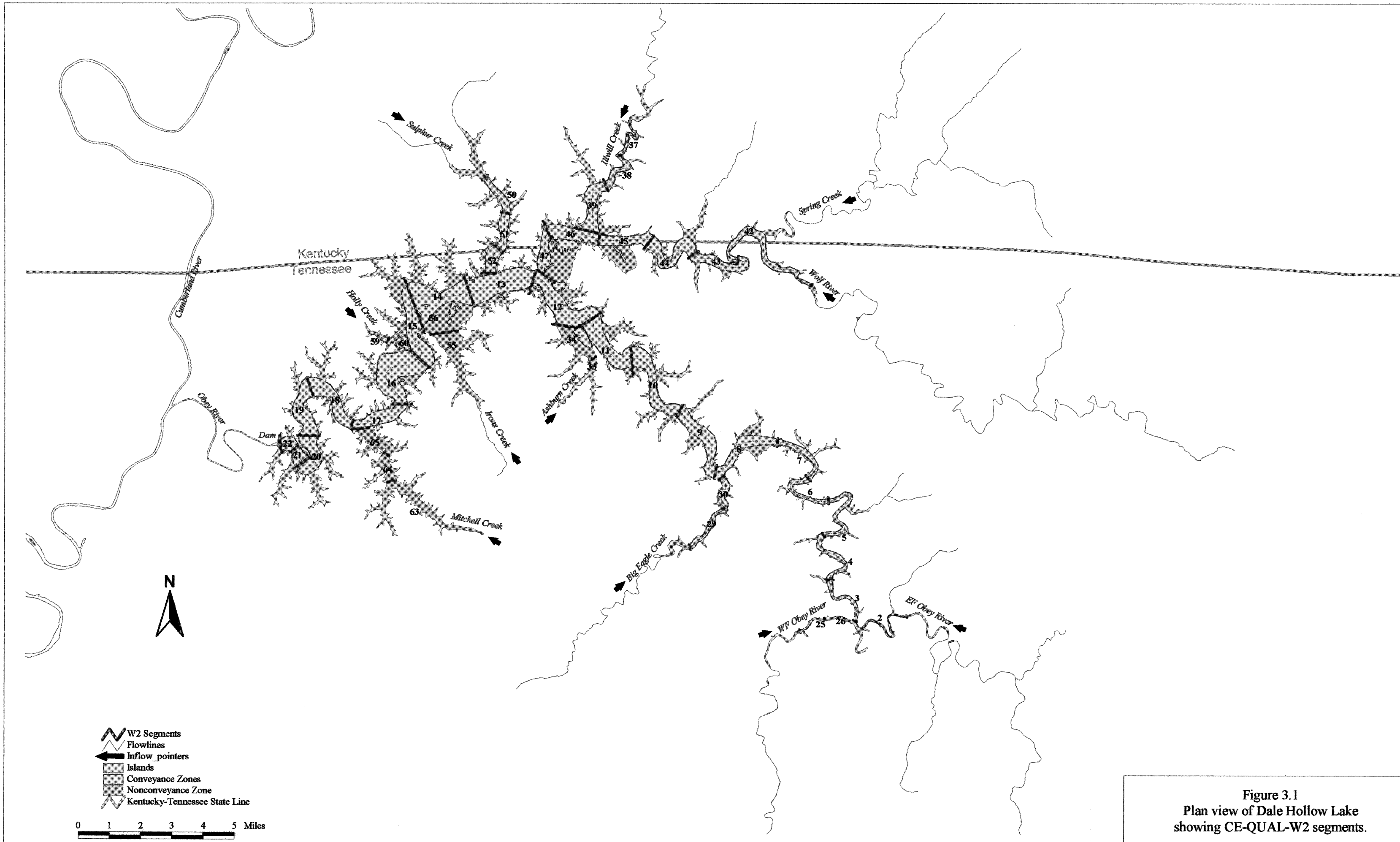


Figure 3.1
Plan view of Dale Hollow Lake showing CE-QUAL-W2 segments.

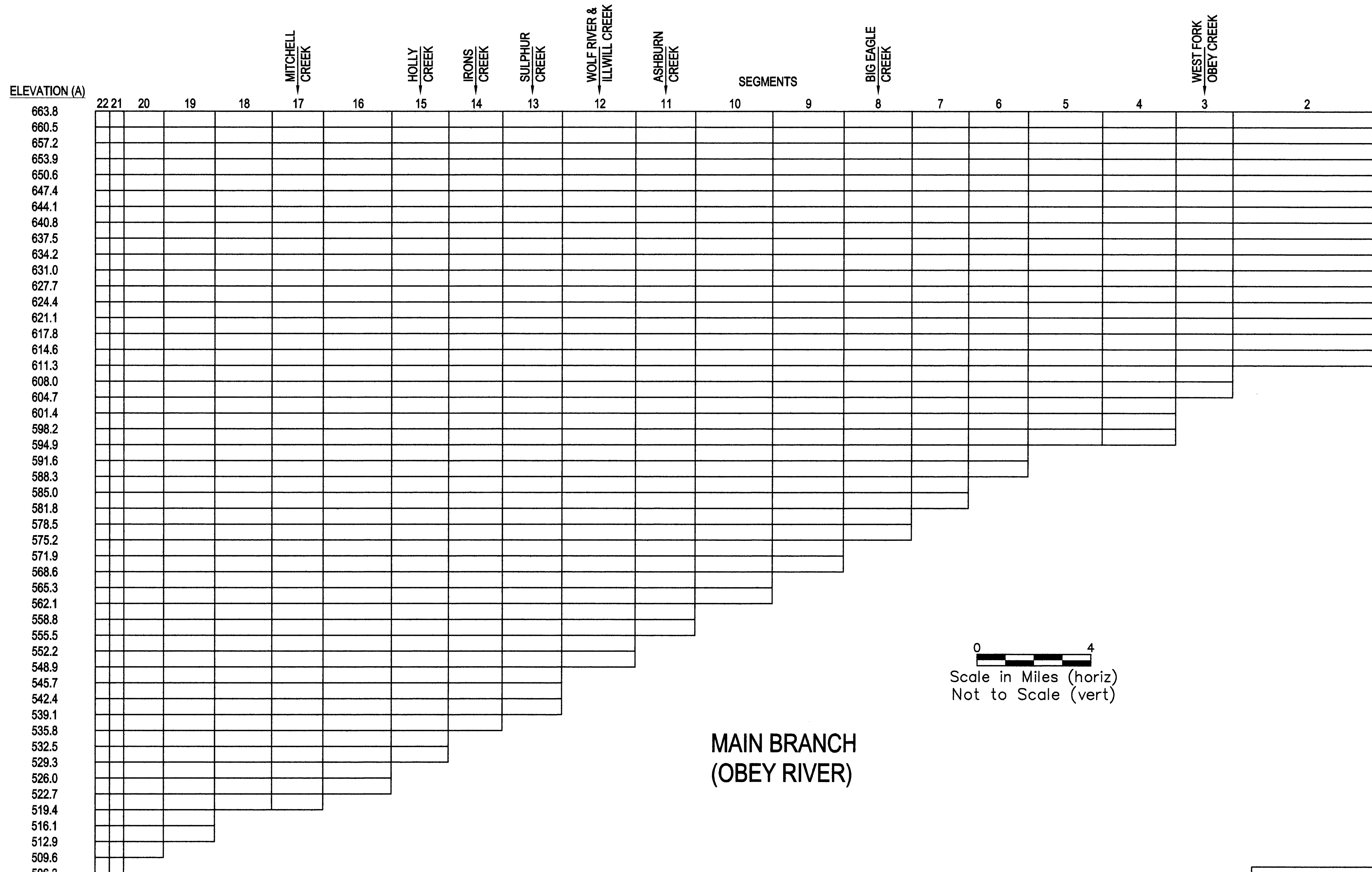


FIGURE 3.2

DALE HOLLOW LAKE
WATER QUALITY MODEL
CE-QUAL-W2 GRID

ftn Associates Ltd.

By: JRB
Date: April 13, 2001
File: 1150FG03.DWG
Project No. 1150-041

Branch 2
(WF Obey River)

26 25

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16

Branch 3
(Big Eagle Creek)

30 29

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24

Branch 4
(Ashburn Creek)

34 33

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28
568.6	30

Branch 5
(Illwill Creek)

39 38 37

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28

Branch 6
(Wolf River)

47 46 45 44 43 42

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28
568.6	30
562.1	32
555.5	34
548.9	36

Branch 7
(Sulphur Creek)

52 51 50

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28
568.6	30
562.1	32

Branch 8
(Irons Creek)

56 55

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28
568.6	30
562.1	32
555.5	34
548.9	36

Branch 9
(Holly Creek)

60 59

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28

Branch 10
(Mitchell Creek)

65 64 63

Layer Bottom El (ft)	Layer
660.5	2
653.9	4
647.4	6
640.8	8
634.2	10
627.7	12
621.1	14
614.6	16
608.0	18
601.4	20
594.9	22
588.3	24
581.8	26
575.2	28
568.6	30
562.1	32
555.5	34
548.9	36
542.4	38
535.8	40
529.3	42
522.7	44
516.1	46

0 4
Scale in Miles (horiz)
Not to Scale (vert)

FIGURE 3.2 (cont)

DALE HOLLOW LAKE
WATER QUALITY MODEL
CE-QUAL-W2 GRID

ftn Associates Ltd.

By: JAB
Date: April 13, 2001
File: 1150FG03.DWG
Project No. 1150-041

3.2 Branches

The model consists of a total of 10 branches and 66 segments, including boundary segments. The main branch extends from the dam up the East Fork Obey River to sampling station 3DAL20009. The other branches that were simulated were West Fork Obey River, Big Eagle Creek, Wolf River, Illwill Creek, Sulphur Creek, Ashburn Creek, Irons Creek, Holly Creek, and Mitchell Creek.

Most of the branches were configured to represent only the zone of conveyance so that movement of water and constituents along the length of the Lake would be simulated as accurately as possible. Inclusion of non-conveyance zones would cause the model to underestimate longitudinal velocities by overestimating the cross sectional area through which the water moves. However, it was also important for the model to accurately represent the total volume of the Lake. This was done by designating three branches as "null embayments" and adding volume to those branches to represent volume in non-conveyance zones throughout the Lake. The three branches designated as null embayments were Ashburn Creek, Irons Creek, and Mitchell Creek. These three branches were selected to be null embayments because they do not have large inflows (based on drainage area). Two other branches (Holly Creek and Sulphur Creek) do not have large inflows, but those branches were not designated as null embayments because they were selected for evaluation of the impacts of houseboats and marinas.

Embayments are modeled as branches with inflows entering their upstream boundary. Null embayments are primarily water storage areas and are modeled as branches with minimal upstream flows. During previous studies it was determined that modeling null embayments as branches with no upstream flow introduced numerous instabilities. In the Dale Hollow Lake configuration, a branch is located on a branch. This configuration is the Illwill Creek embayment, which flows into the Wolf River embayment. Based upon our experience with CE-QUAL-W2, and difficulties previously encountered calibrating the model with a relatively large number of branches, specific attention was devoted to this branch connection. No anomalies were identified at this connection during the temperature calibration process, nor were any problems encountered with any type of noniterative solution scheme that has in past applications affected calibration at branch intersections.

Main branch and embayment cells represent the conveyance channel of the main channel and important tributaries (refer to Figure 3.1). The six embayments were modeled at West Fork Obey River, Big Eagle Creek, Wolf River, Illwill Creek, Sulphur Creek, and Holly Creek. The three null embayments were modeled at Ashburn Creek, Irons Creek, and Mitchell Creek. Spring Creek is included in the Dale Hollow Lake model as a tributary.

The main branch and embayment cells constitute the bulk of the volume included in the model. Volume was added to null embayments as necessary to ensure that the total volume of the model corresponds to the District's elevation-volume curve. The embayments consist of 2-22 segments. The null embayments have 2-3 segments. The null embayments represent the volume from non-conveyance areas throughout the lake.

Holly Creek represents a small embayment and Sulphur Creek represents a medium to large embayment. Segment lengths in these prototype embayments are small in order to allow simulation of the effects of marinas and houseboats on water quality.

3.3 Average Widths

Cell widths were developed based on transects collected by the District and supplemented with information from fishing maps with contours (Westmorland 1980). Transects were converted into HEC format and input to GEDA, which calculated volumes at specified elevations. Average widths for each cell were calculated from these volumes. The volumes of the model bathymetry have been checked against the District elevation-volume curve.

The null embayment volumes were calculated based on the difference between the total volume per layer accounted for in the main branch and embayments and the total volume per elevation (layer) reported by the District. This excess volume was distributed between the three null embayments based on the following criteria:

- Starting at the bottom of the reservoir and working upwards, volume was added to each null embayment starting at the approximate bottom elevation at the mouth of that null embayment (e.g., volume was added to the Mitchell Creek null embayment at lower elevations than for the Ashburn Creek null embayment).

- Each time volume was assigned to a cell in a null embayment, the resulting cell width was checked to make sure that it was as least as wide as the cells immediately below and immediately upstream (i.e., cell widths for null embayments were not allowed to decrease in the downstream direction or in the upward direction).

The cell widths were then calculated by dividing the applied volume by the segment length and the layer height.

Previous experience with the CE-QUAL-W2 model has shown that small cell widths along the bottom of the reservoir can cause numerical instabilities during simulation. Therefore, all cell widths that were less than 20 m were added to the next available cell width to keep the integrity of the volume as well as preserve the minimum cell width of 20 m. The maintenance of the minimum cell width of 20 m has been used successfully in previous studies.

Figure 3.3 shows a comparison of the model elevation-volume curve to the District's elevation-volume curve for Dale Hollow Lake. Overall the model volume nearly matches the reported reservoir volume. The model elevation-volume curve tends to slightly underestimate the volume at elevations above 585 ft, msl; however, the model volume was revised at elevation 663.8 ft, msl to maintain the overall value of the capacity curve.

3.4 Segment Orientation

Segment orientation was estimated to the nearest 10 degrees from the GIS map provided by the District (Figure 3.1) using a protractor. Segment orientation and wind direction affect hydrodynamics through longitudinal surface velocities and wind-generated shear stresses.

3.5 Outlet Configuration

The model includes discharge sources. Turbine releases were modeled as a point sink with a withdrawal centerline elevation of approximately 584 ft, msl. This elevation was modified from the District's SELECT simulations (Memo to Files dated 13 November 98) (Appendix E). Specific discussion of the turbine outlet configuration is located in Section 4.2.6. The model also includes withdrawal of water for the Dale Hollow National Fish Hatchery as a point sink. The withdrawal rates for the study years were provided by the District. The average withdrawal rate prior to a 1993 expansion was about 7200 gallons per minute (16 cfs). For the projection simulations, the configuration of this withdrawal in the model was changed to reflect current conditions (after the 1993 expansion). The average withdrawal rate during the study years of 1973, 1988, and 1991 was 17 cfs. The elevation of the fish hatchery withdrawal is 571.5 ft, msl. Spillway releases (tainter gates) were modeled as a line sink 400 ft wide having a withdrawal elevation of 651 ft, msl, as described on the Dale Hollow dam plans. The service unit was modeled as a point sink with a withdrawal centerline elevation of 556.5 ft, msl, as per the District (Brown 2000). Sluice releases were also modeled as a point sink with a withdrawal centerline elevation of 537 ft, msl. The centerline elevation of the sluice gates was also taken from the Dale Hollow dam plans as provided by the District.

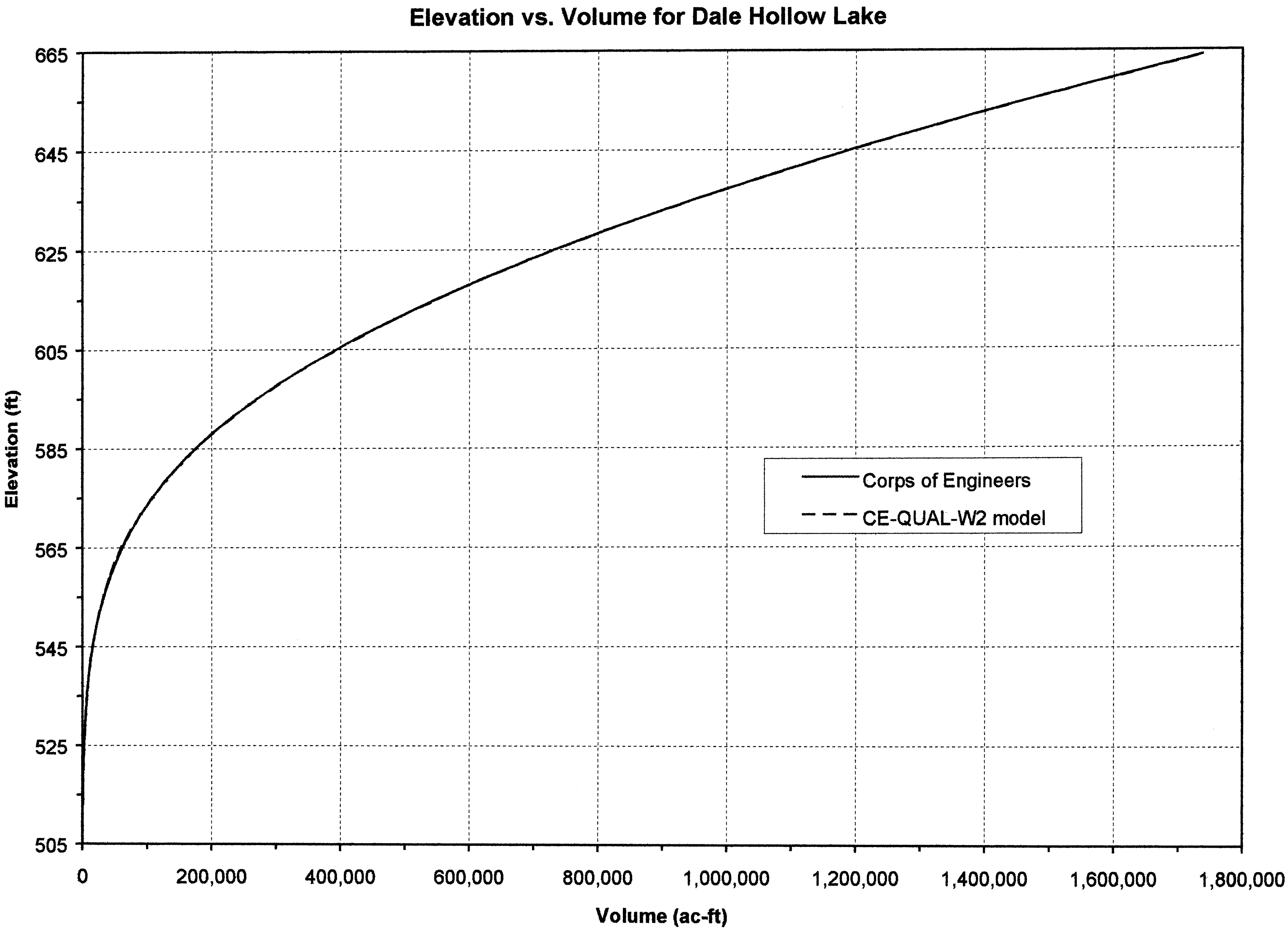


Figure 3.3. Comparison of model elevation capacity curve for Dale Hollow Lake

4.0 MODEL CALIBRATION

The CE-QUAL-W2 model was calibrated by simulating three different years: 1973, 1988, and 1991. These three years were selected to represent a range of hydrologic conditions, including wet (1973), dry (1988), and average (1991) hydrologic conditions. The method and rationale used to select these years is presented in Appendix F.

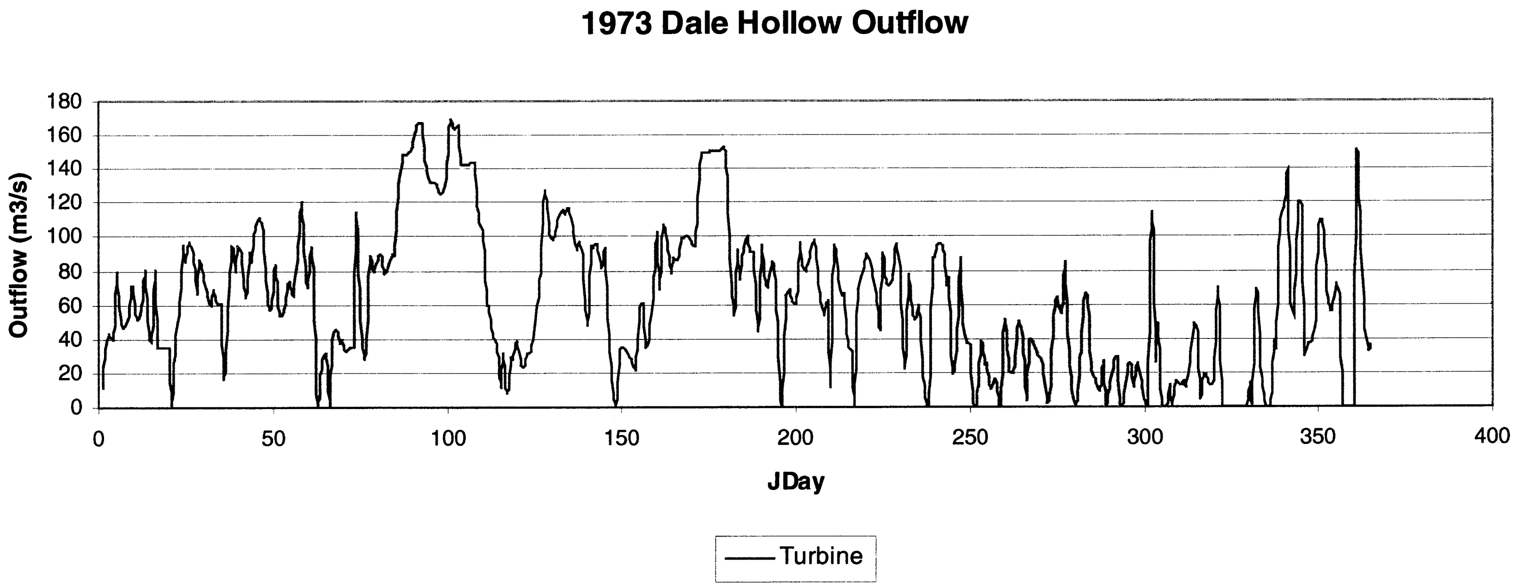
The calibration process was similar to that recommended in the CE-QUAL-W2 User's Manual (Cole and Buchak, 1994). Once the reservoir bathymetry was developed, the water budget was checked by comparing computed and measured water surface elevations over time. Temperature was then calibrated to reflect heat budget computations and reservoir hydrodynamics. Temperature calibration was followed by calibration of the water quality parameters. Temperature and water quality parameters were calibrated by comparing measured data with model output for each parameter and then modifying model coefficients based on physiochemical phenomena to minimize the difference. Printouts of the CE-QUAL-W2 control files for the calibrated Dale Hollow Lake models (1973, 1988, and 1991) are included as Appendix G.

4.1 Water Budget

For each study year, a daily water budget was developed based on inflows from the watershed and outflows at the dam. Loss of water from evaporation was not simulated in the model and was therefore implicit in the water budget.

4.1.1 Outflows

The District provided hourly outflows during the study years for the turbines, sluice, service unit, fishery withdrawal, and spillway. Each of these five outflows was specified separately in the model (i.e., they were not lumped together as a single outflow). The hourly outflows provided by the District were used as model input with no adjustments. Daily averages of the total outflow are shown in Figures 4.1 – 4.3.



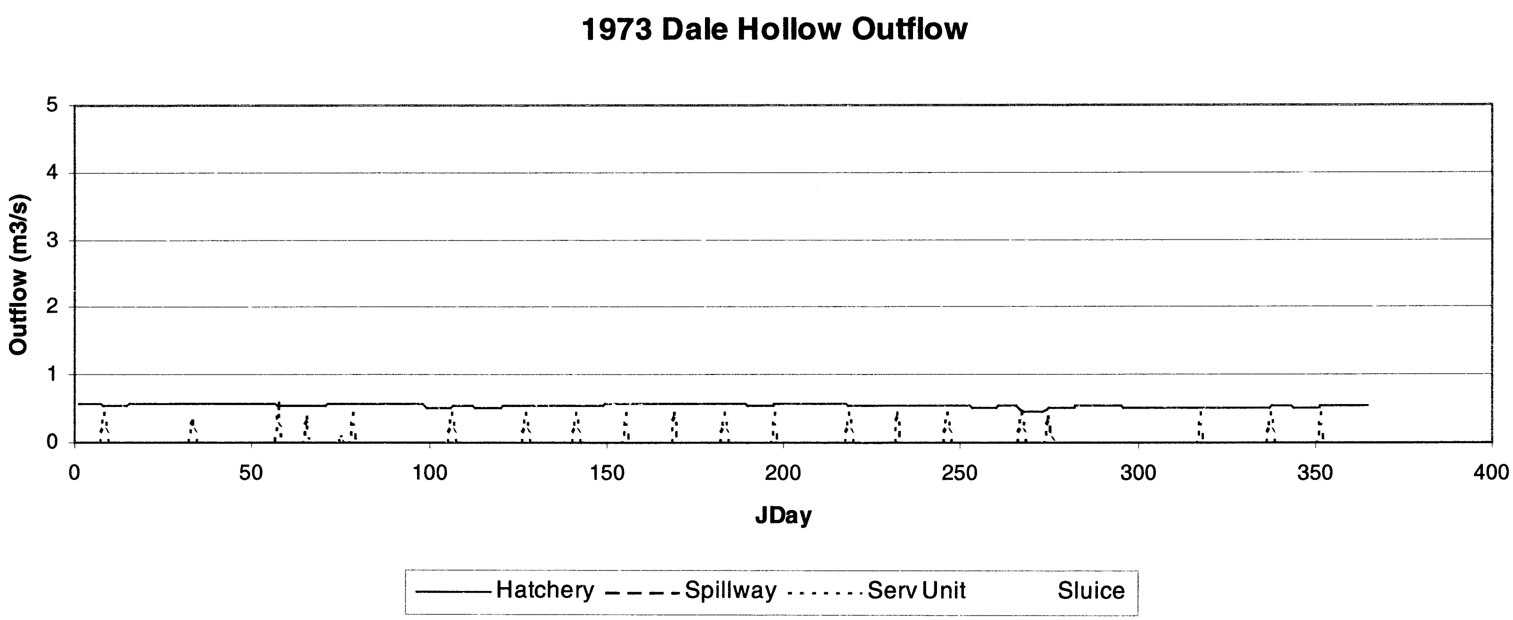
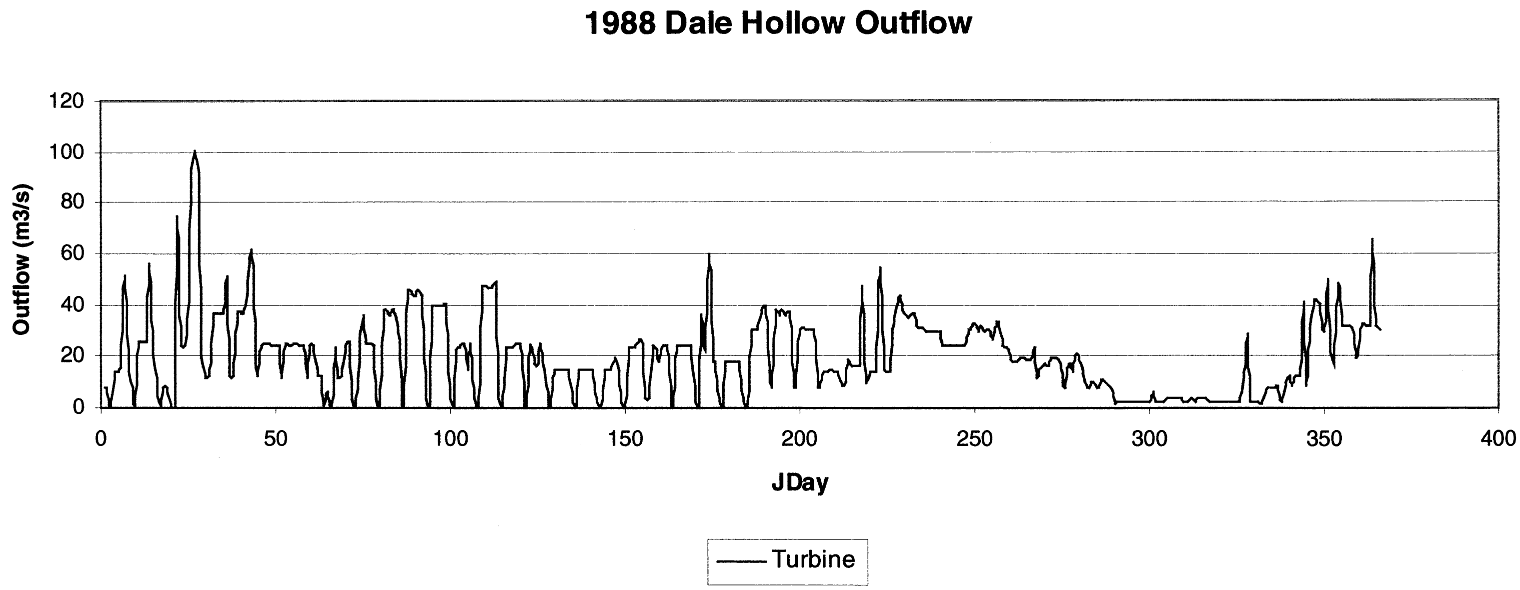


Figure 4.1. 1973 Outflows.



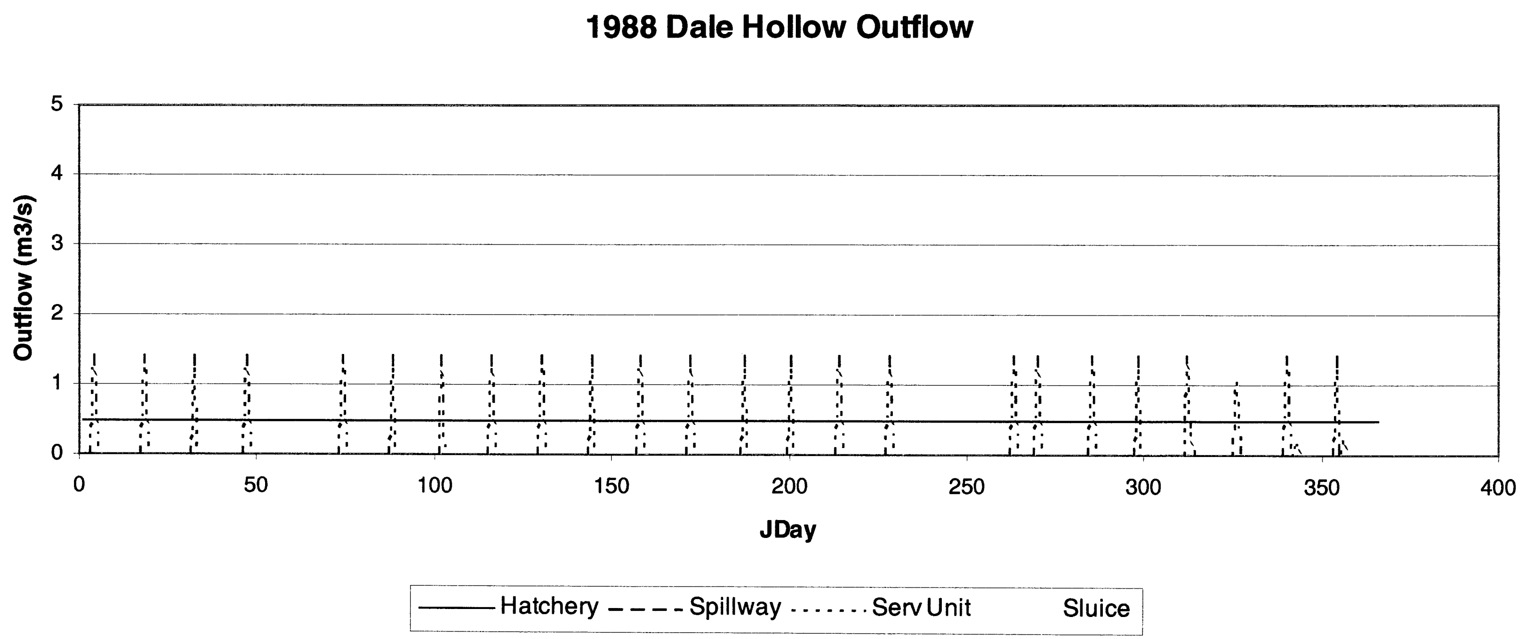
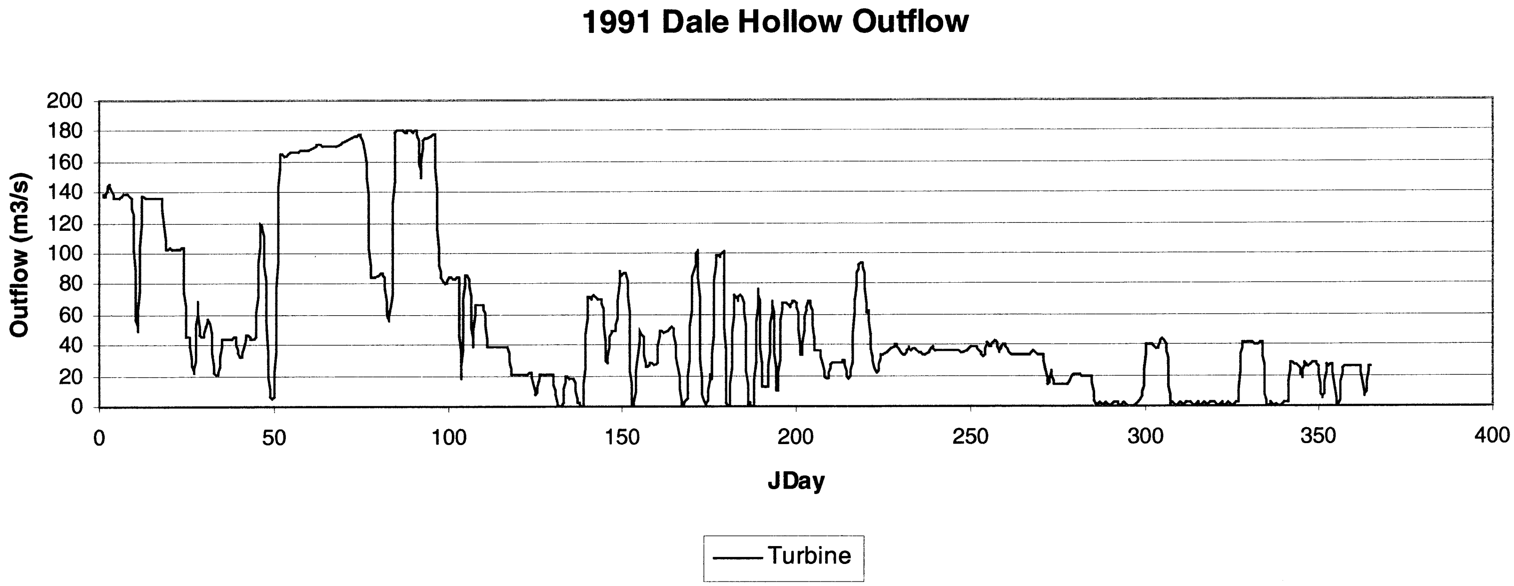


Figure 4.2. 1988 Outflows



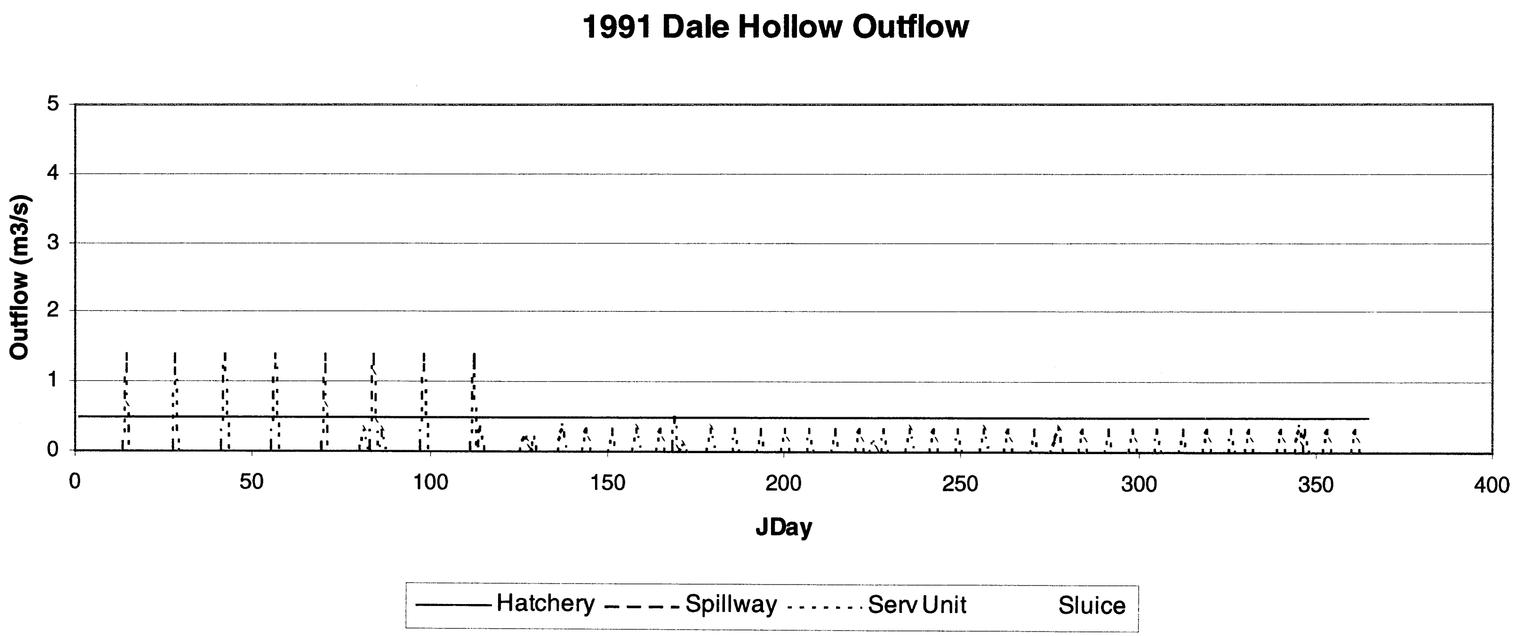


Figure 4.3. 1991 Outflow

4.1.2 Inflows

Inflows that were specified in the model were East Fork Obey River, West Fork Obey River, Wolf River, Big Eagle Creek, Spring Creek, Illwill Creek, Sulphur Creek, Ashburn Creek, Holly Creek, Irons Creek, and Mitchell Creek. These streams account for approximately 72% of the total drainage area at the dam. In addition, a distributed inflow to the main branch was included in the model to account for smaller tributaries and direct inflow to the lake. Flow data that were available for the study years consisted of daily flows measured by the USGS for East Fork Obey River (gage number 03414500) and Wolf River (gage number 03416000), along with the District's calculated total inflows to the lake.

Table 4.1. Drainage areas for inflows to Dale Hollow Lake.

Watershed	Drainage Area (square miles)	% of Total
East Fork Obey River	202	21.6%
West Fork Obey River	115	12.3%
Wolf River	106	11.3%
Big Eagle Creek	45	4.8%
Spring Creek	60	6.4%
Illwill Creek	52	5.6%
Sulphur Creek	25	2.8%
Ashburn Creek	20	2.1%
Irons Creek*	20	2.1%
Mitchell Creek*	20	2.1%
Holly Creek*	10	1.1%
Distributed Tributary	260	27.8%
Total	935	100.0%

* Note: Estimated drainage area

4.1.2.1 Inflow Rates for Streams with USGS Gages

Measured flows from the USGS were used to specify inflows in the model for East Fork Obey River, West Fork Obey River, and Wolf River. The USGS measured flows were considered to be the most accurate estimations of the actual amounts of water flowing into the Lake through those streams. For East Fork Obey River and Wolf River, published daily flows were available for water years 1944 – 1991 (i.e., ending in September 1991). For October through December 1991, provisional daily flow data were obtained for these two gages from a

CD-ROM containing USGS data (Hydrosphere 2000). These published and provisional flows for East Fork Obey River and Wolf River were specified as model input with no adjustments. These values are plotted in Figures 4.4 and 4.5.

For West Fork Obey River (USGS gage number 03415000), the gage was discontinued in 1971. To determine if flows on the East Fork Obey River could be used to estimate flows on the West Fork Obey River during the study years, average flows per unit area were compared for these two gages. For water years 1944 – 1971, the average flow per unit of contributing drainage area were 1.97 cfs/mi^2 for East Fork Obey River and 1.95 cfs/mi^2 for West Fork Obey River. Because these flows per unit area were similar, the daily flows for the West Fork Obey River during the study years were estimated by multiplying each daily flow for the East Fork Obey River by the ratio of the drainage areas of these two streams. These estimated flows were used as model input for the West Fork Obey River. These values are plotted in Figure 4.6.

4.1.2.2 Inflow Rates for Other Streams

For the other nine inflows to the lake (Big Eagle Creek, Spring Creek, Illwill Creek, Sulphur Creek, Ashburn Creek, Irons Creek, Holly Creek, Mitchell Creek, and the distributed tributary), daily flows were estimated by determining the total inflow to the lake on each day and then subtracting the inflow for the three gaged streams. To determine the total inflow to the lake, the District's calculated total inflows were examined. Because these inflows are calculated based on outflow and change in storage in the lake, they represent total "net" inflow to the reservoir, which is equivalent to actual inflow minus evaporation. Pan evaporation rates during June through August are typically about 0.20 in/day, which is equivalent to a loss of 153 cfs over the surface of the lake. This is based on long term averages of pan evaporation data at Cordell Hull Lock and Dam and Wolf Creek Dam and assumes a pan coefficient of 0.7 and an approximate surface area of 26,000 acres. Because the loss from evaporation often exceeds the amount of actual inflow to the lake during the late summer and early fall, the calculated inflows are often less than zero. To be able to use these data as upstream flows in the CE-QUAL-W2 model, the HECUPD program (USACE, 1991) was used to bring the negative values up to a user-specified minimum value while preserving the total volume of flow over the whole simulation period. The minimum flows specified in HECUPD were the sums of the lowest daily flows for the three gage streams; these were 28 cfs in 1973, 14 cfs in 1988, and 25 cfs in 1991.

Using the District's adjusted calculated total inflows and the summation of the measured inflows for the three gaged streams, the total inflow to the lake was based on one of the following two options:

1) If the difference between the District calculated inflows and the USGS gages summed, was greater than 1 cfs, this difference was applied to the nine ungaged tributaries based on their respective drainage areas.

2) If the difference between the District calculated inflows and the USGS gages summed was equal to or less than 1 cfs, 1 cfs was distributed among the nine ungaged tributaries based on their respective drainage areas.

This resulted in all of the daily total inflows to the lake being at least 1 cfs more than the daily sum of the flows for the three gaged streams. This also resulted in an increase in the annual volume of inflow to the lake so that the water budget was no longer balanced. To alleviate this, the adjusted calculated total inflows were decreased slightly by multiplying them by a scaling factor so that the water budget would still balance on an annual basis. The scaling factors used were 0.99 in 1973, 0.96 in 1988, and 0.99 in 1991.

To specify the model input for each day for the nine inflows that were not gaged streams, the sum of the three gaged inflows was subtracted from the total inflow to the lake and the remaining flow was divided among the nine inflows in proportion to their drainage areas. The values used as model input for these nine inflows are plotted in Figures 4.7 – 4.15.

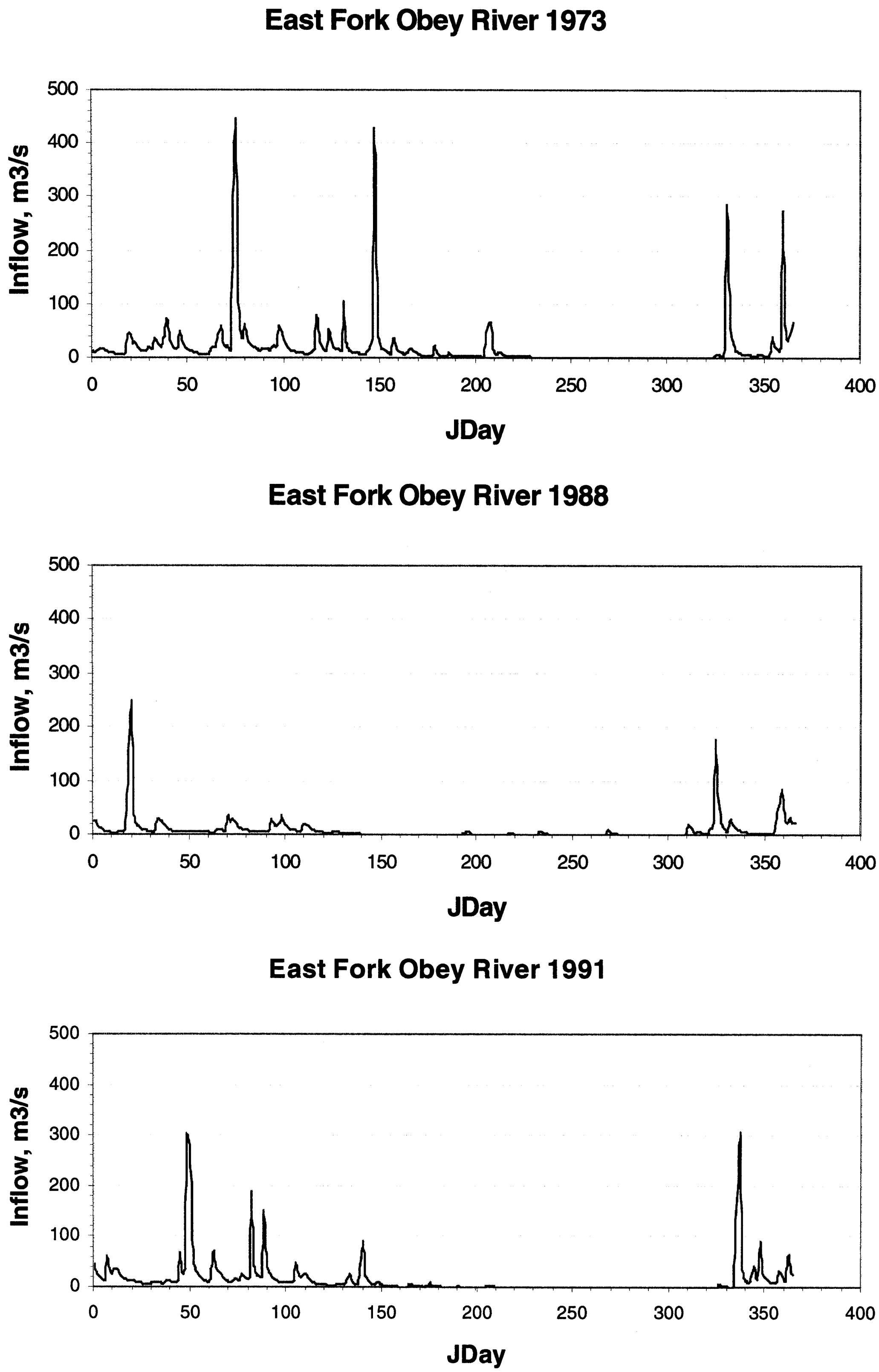


Figure 4.4. Inflows from East Fork Obey River.

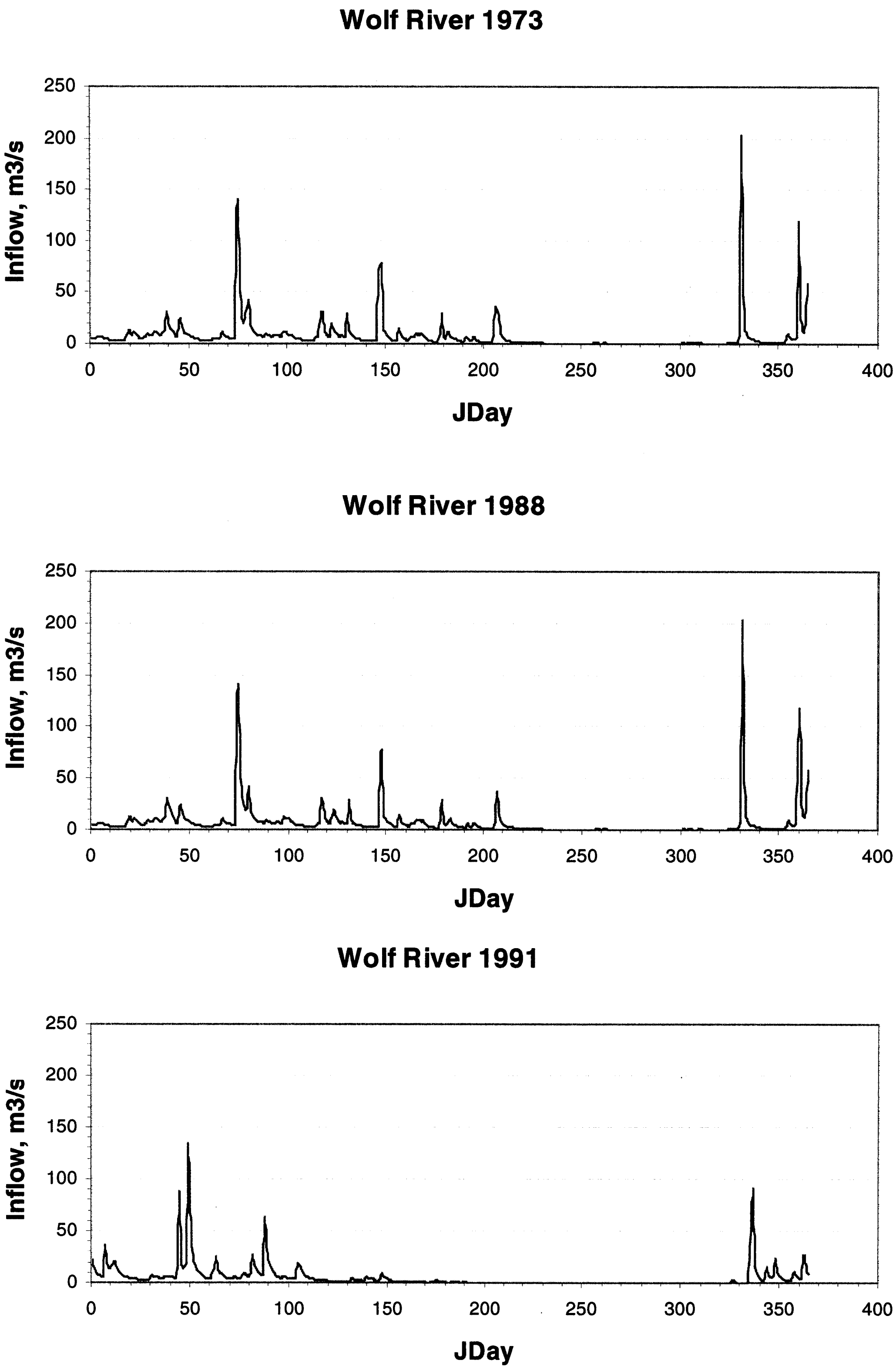


Figure 4.5. Inflows from Wolf River.

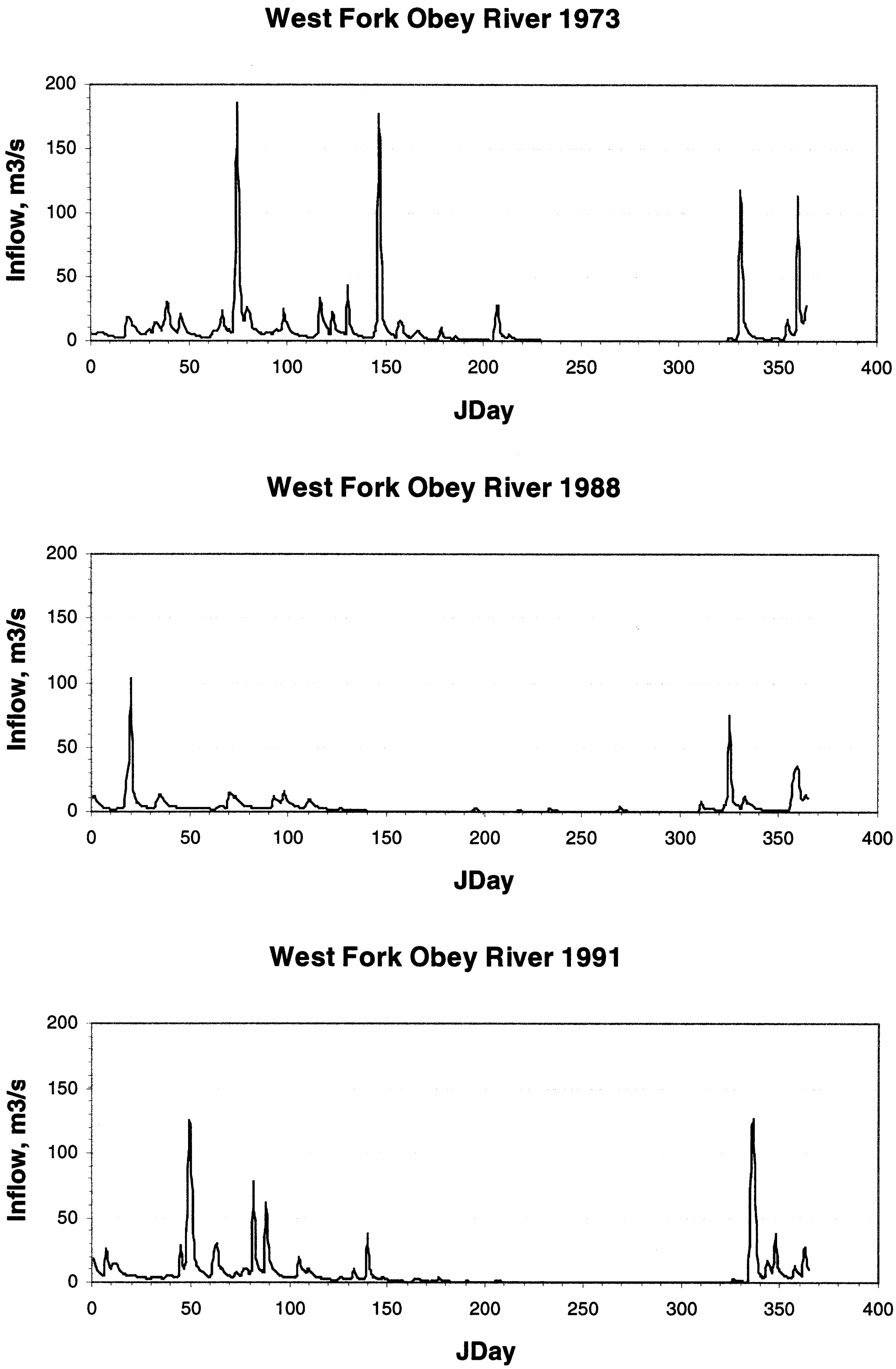


Figure 4.6. Inflows from West Fork Obey River.

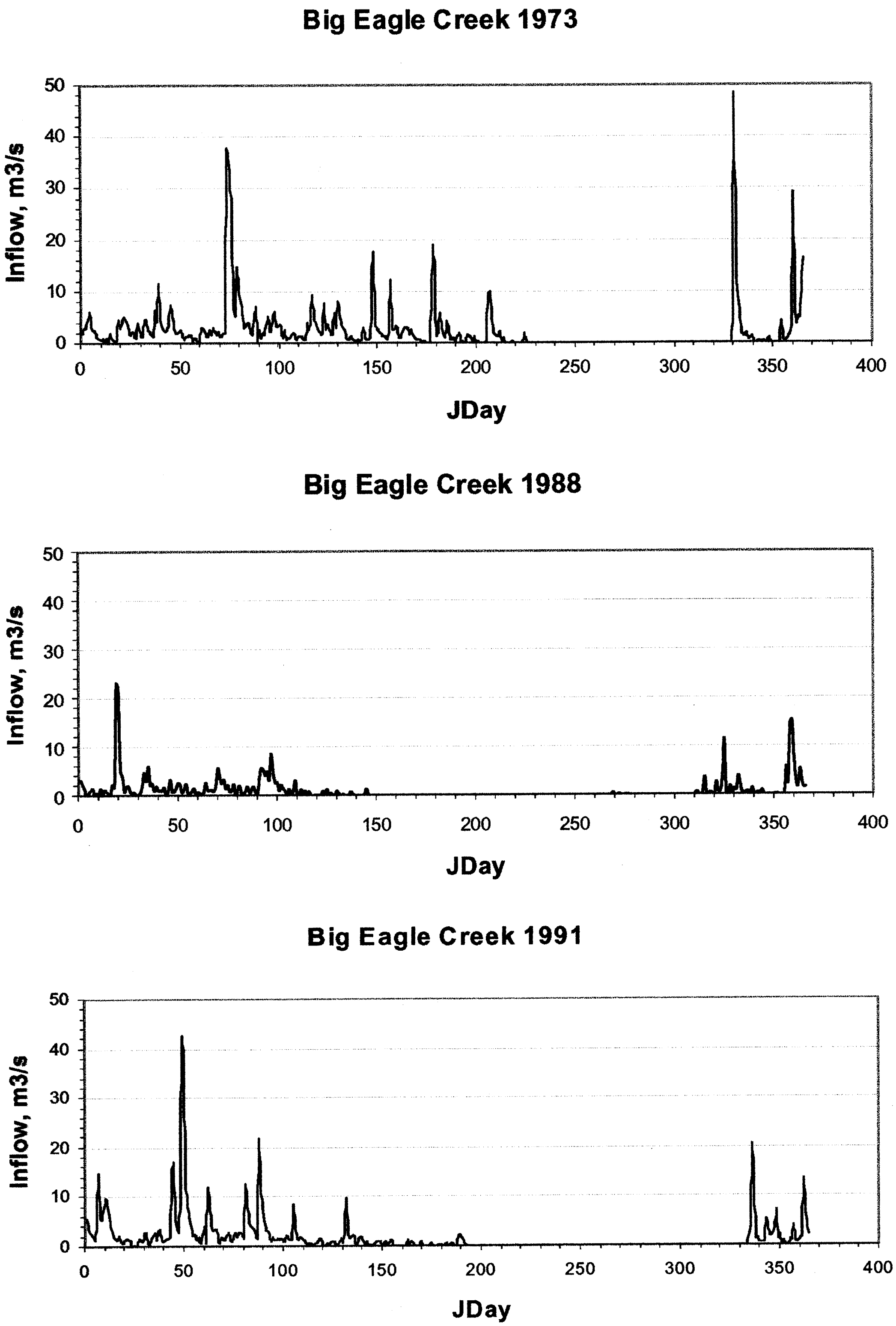


Figure 4.7. Inflows from Big Eagle Creek.

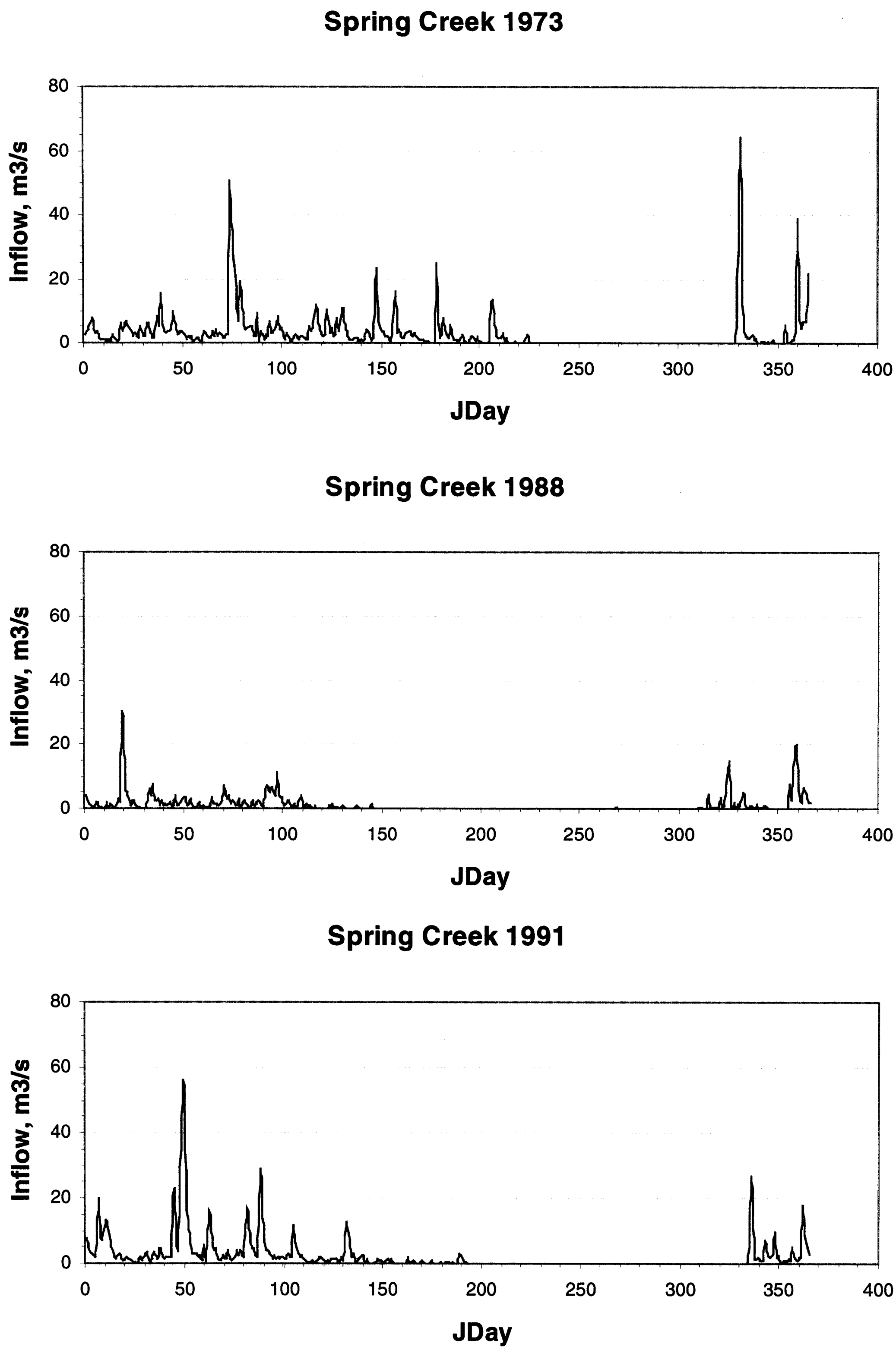


Figure 4.8. Inflows from Spring Creek.

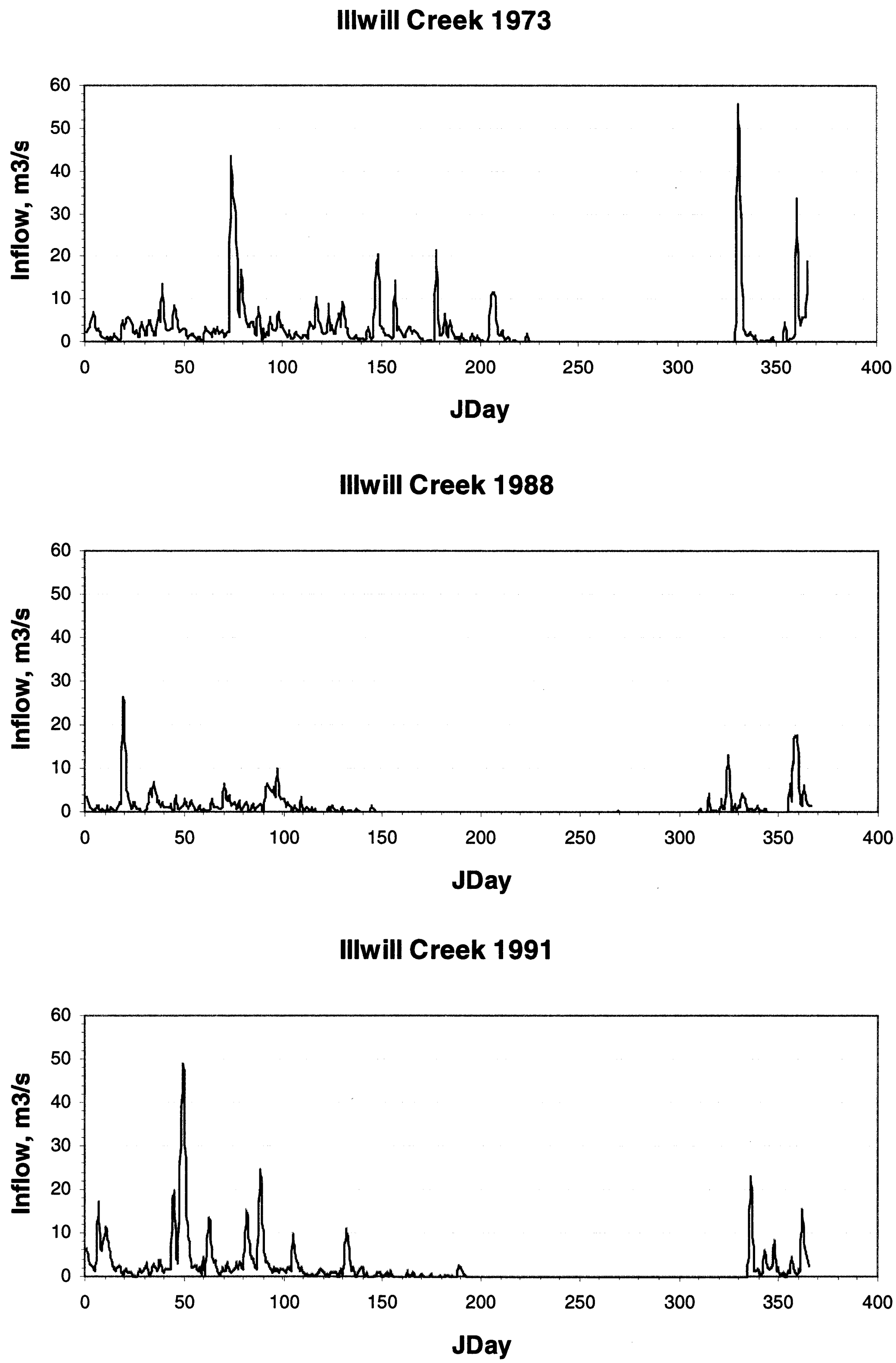


Figure 4.9. Inflows from Illwill Creek.

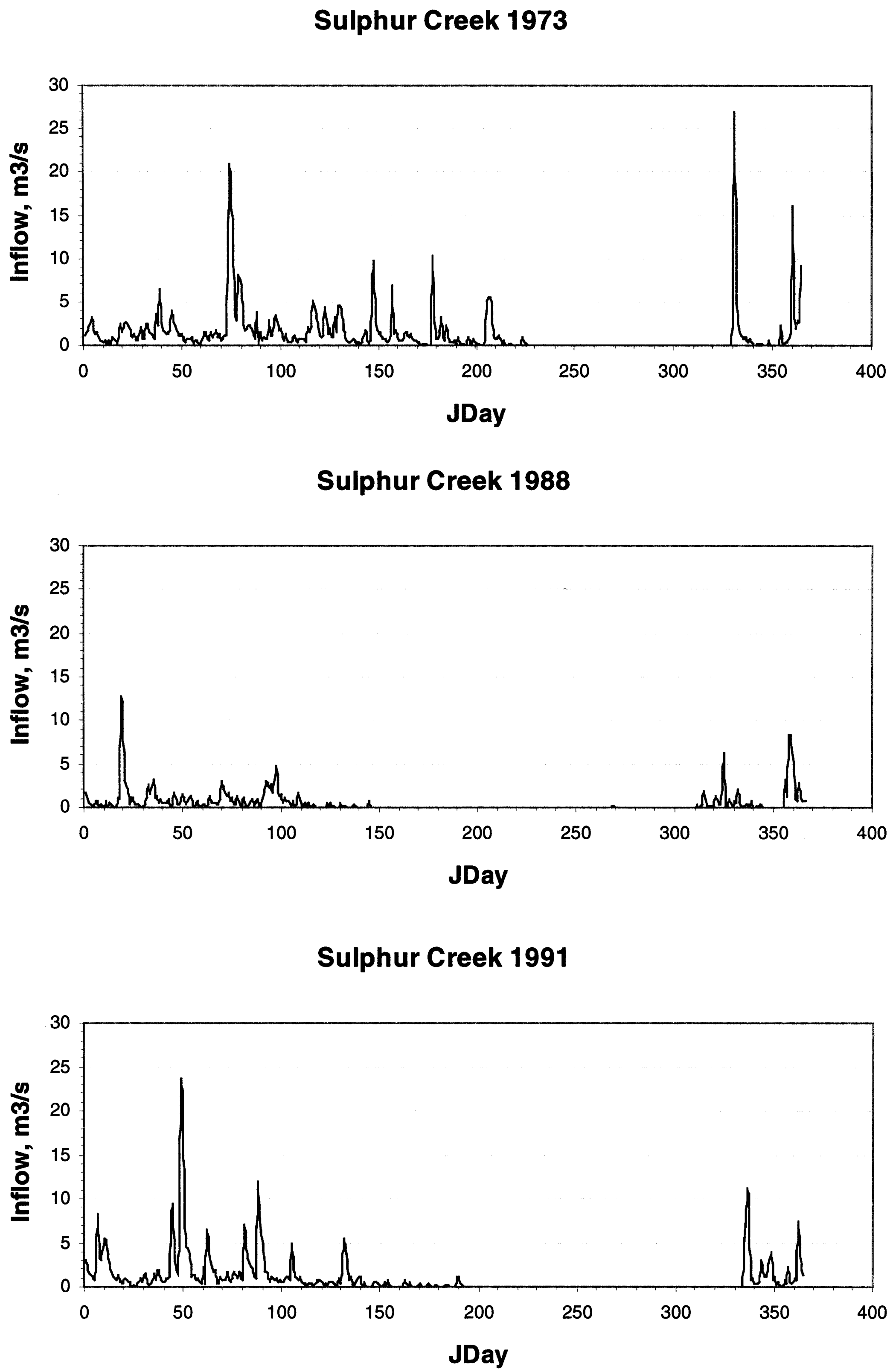


Figure 4.10. Inflows from Sulphur Creek.

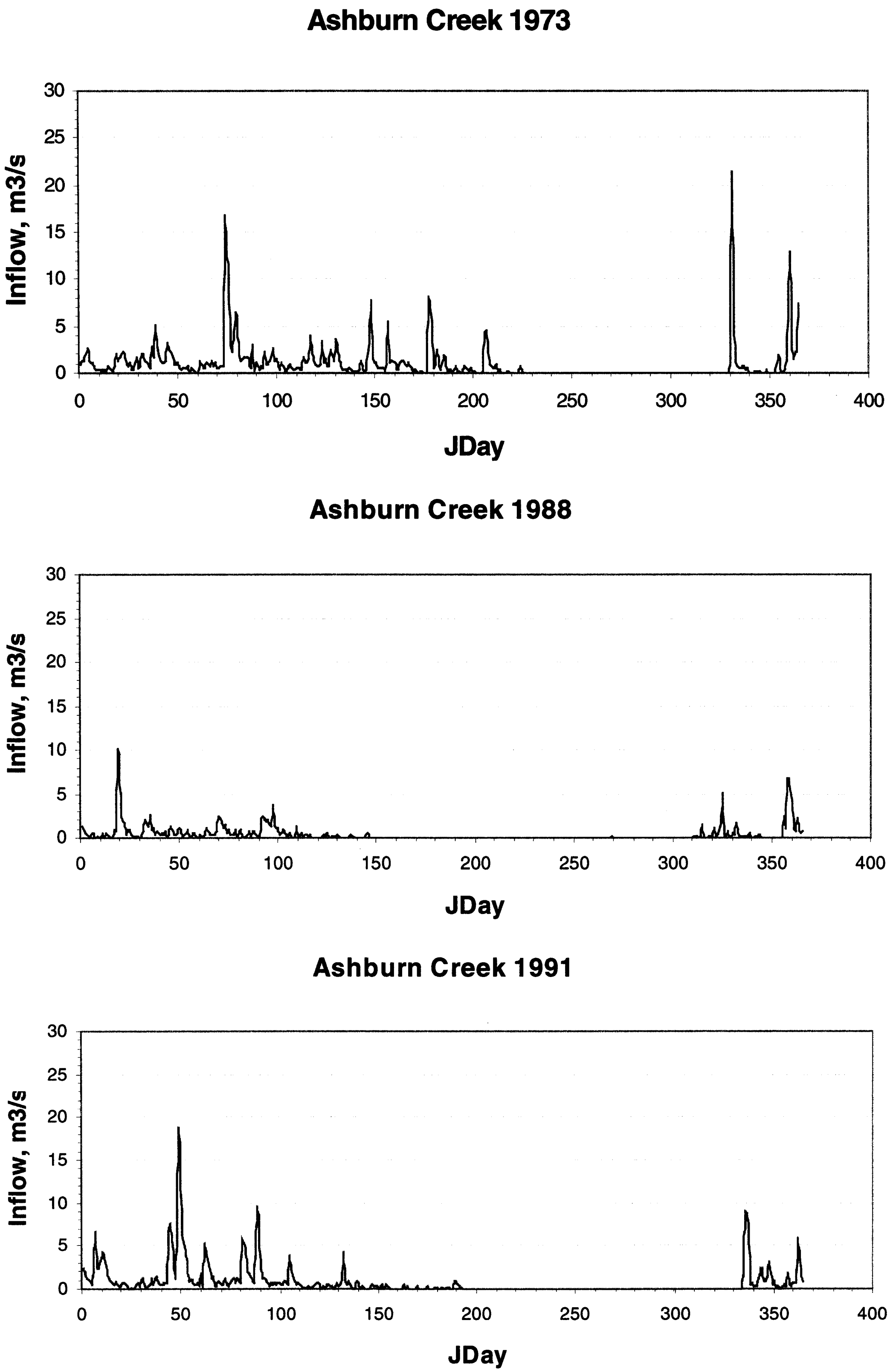


Figure 4.11. Inflows from Ashburn Creek.

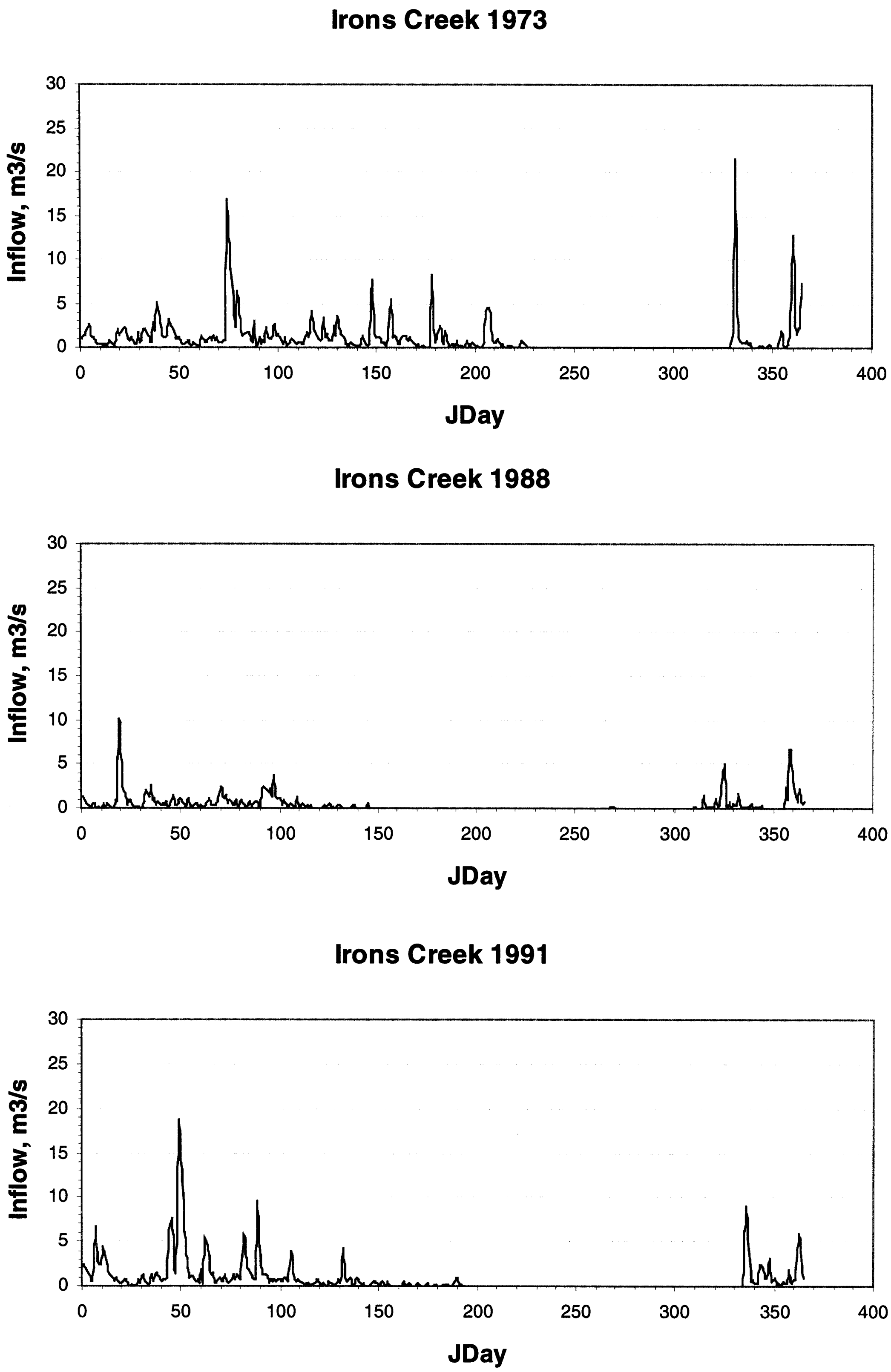


Figure 4.12. Inflows from Irons Creek.

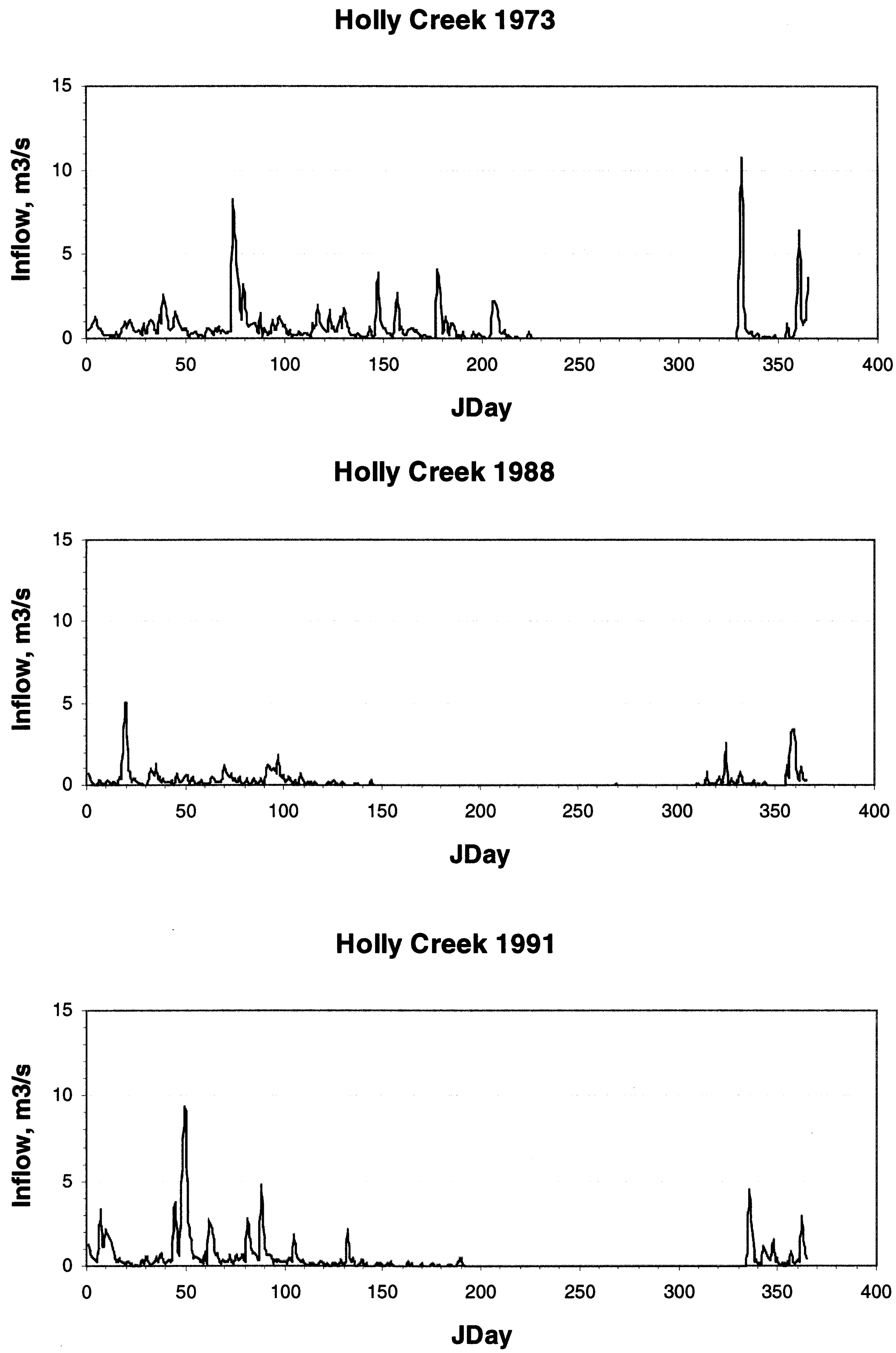


Figure 4.13. Inflows from Holly Creek.

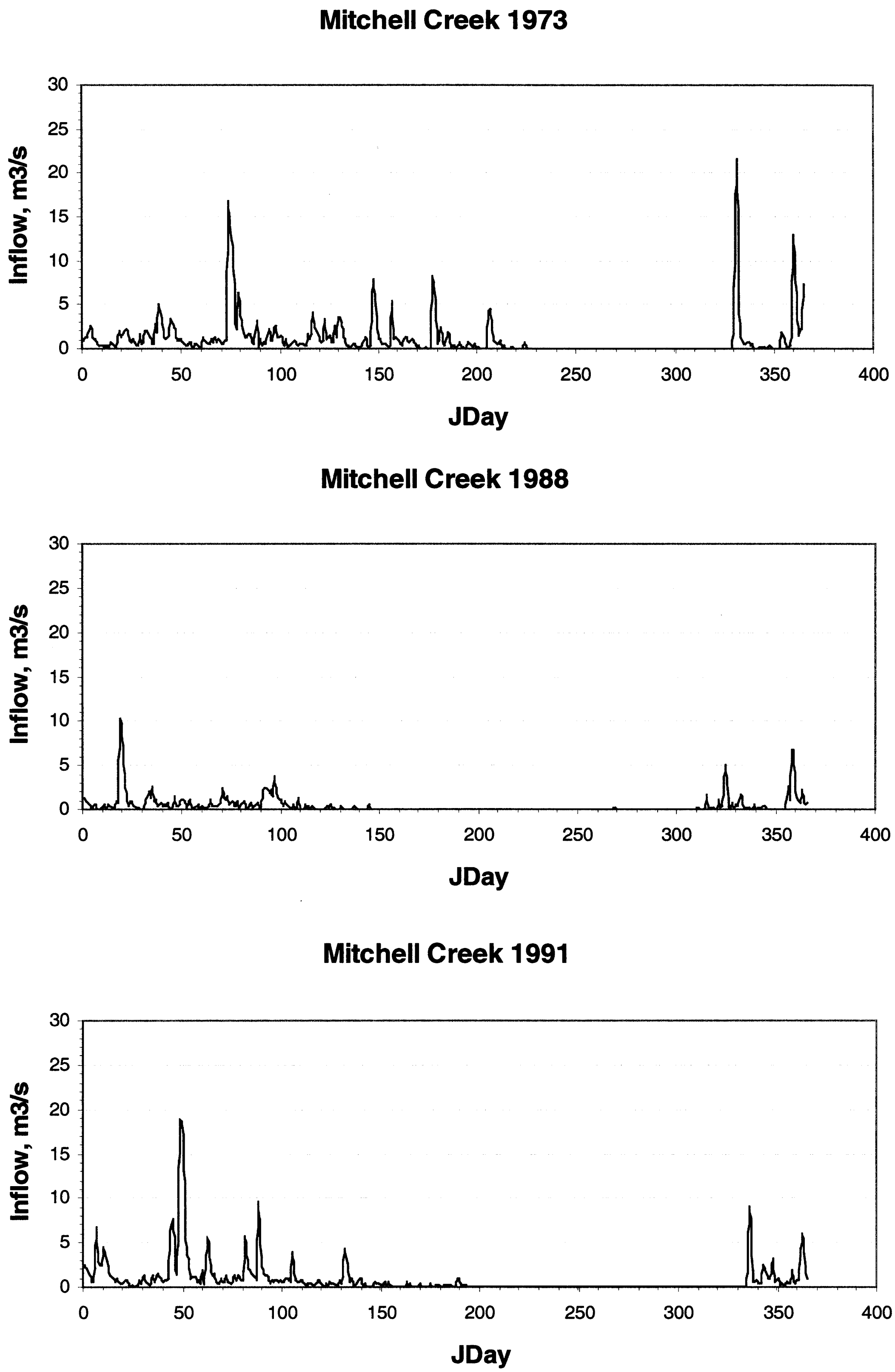


Figure 4.14. Inflows from Mitchell Creek.

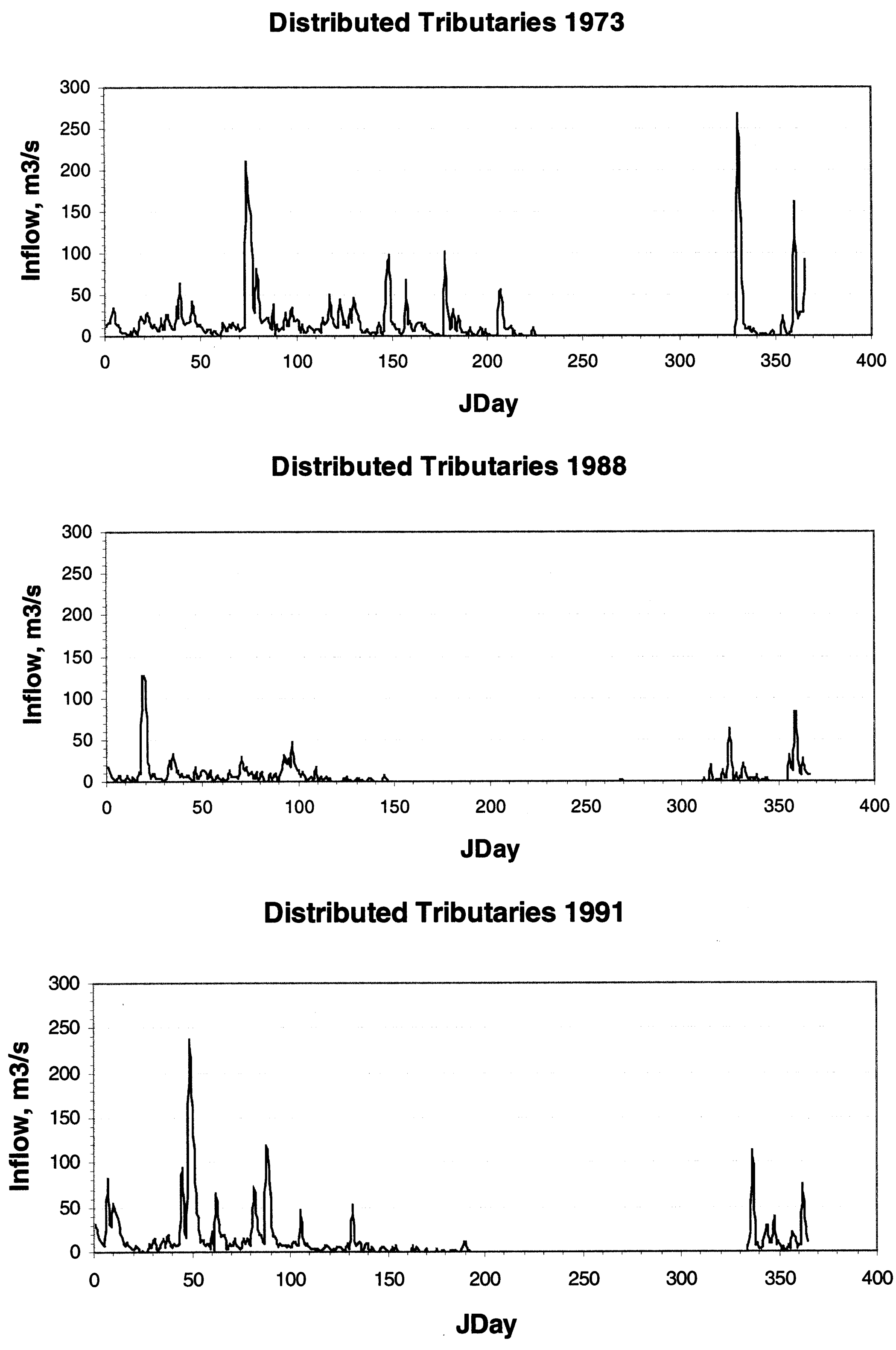


Figure 4.15. Inflows from Distributed Tributaries.

4.1.3 Evaluation of Water Budget

In order to verify that the water budget was balanced, the water surface elevations predicted by the model, using the adjusted inflows described in Section 4.1.2, were compared to the observed water surface elevations provided by the District (Figures 4.16 – 4.18). The predicted water surface elevations were within less than 1.0 ft of the observed values for all study years. The average difference between predicted and observed water surface elevations was less than 0.5 ft, as summarized in Table 4.2.

Table 4.2. Verification of balance of water budget for each study year.

Year	Difference between predicted and observed elevations (ft)	
	Greatest difference	Average difference
1973	0.80	0.41
1988	0.62	0.19
1991	0.71	0.11

4.2 HYDRODYNAMIC AND TEMPERATURE CALIBRATION

Reservoir hydrodynamics play an important role in determining reservoir water quality. The interaction of these hydrodynamic processes with external forcing functions (e.g., wind speed and direction, solar radiation, evaporation, inflow, and outflow) is evaluated during calibration to identify those processes most influential in controlling the hydrodynamics of Dale Hollow Lake. Water temperature is well recognized as a primary factor influencing hydrodynamics; therefore, calibration for both is an interdependent process.

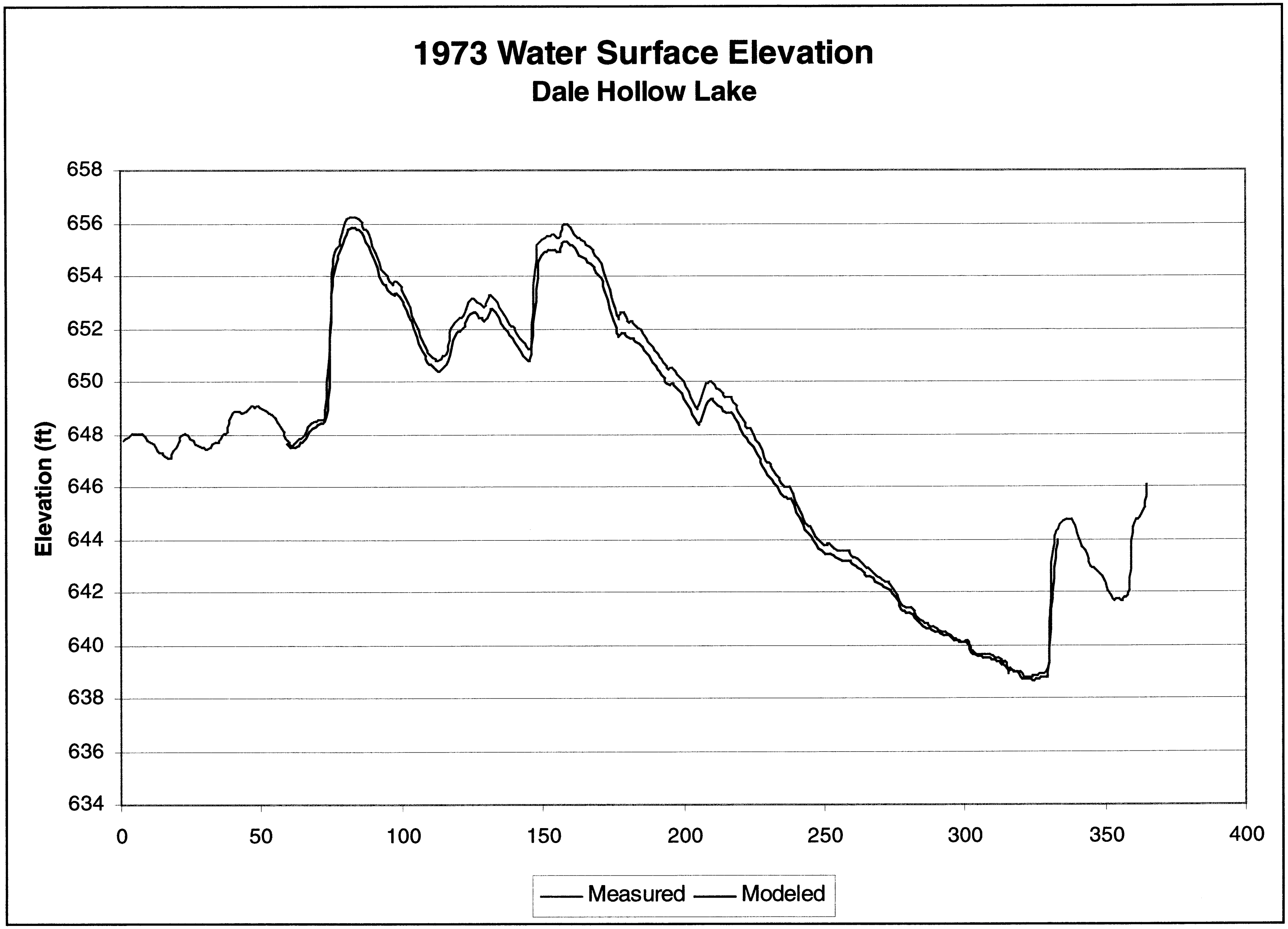


Figure 4.16. 1973 Measured and modeled Dale Hollow Lake pool elevation.

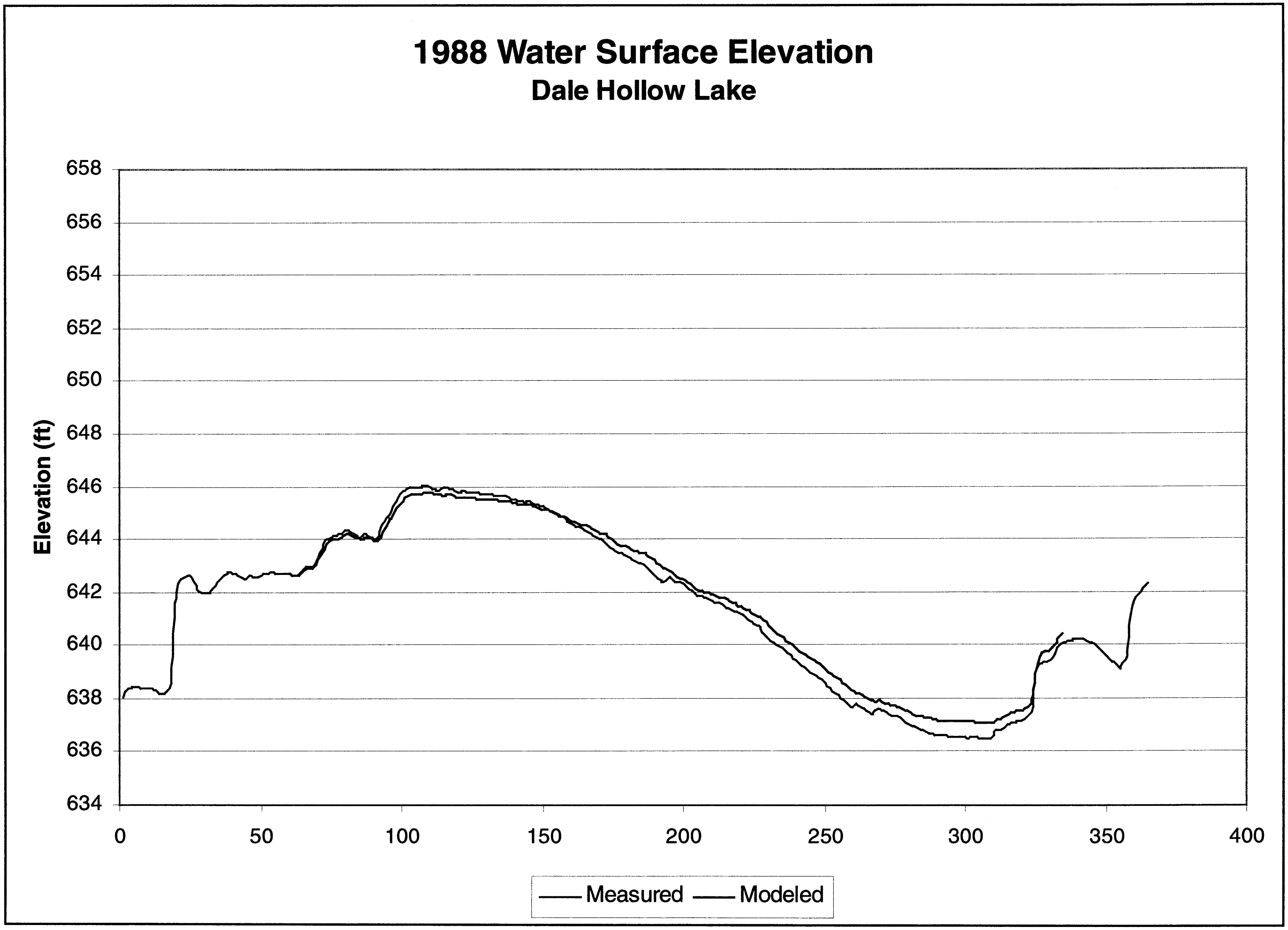


Figure 4.17. 1988 Measured and modeled Dale Hollow Lake pool elevation.

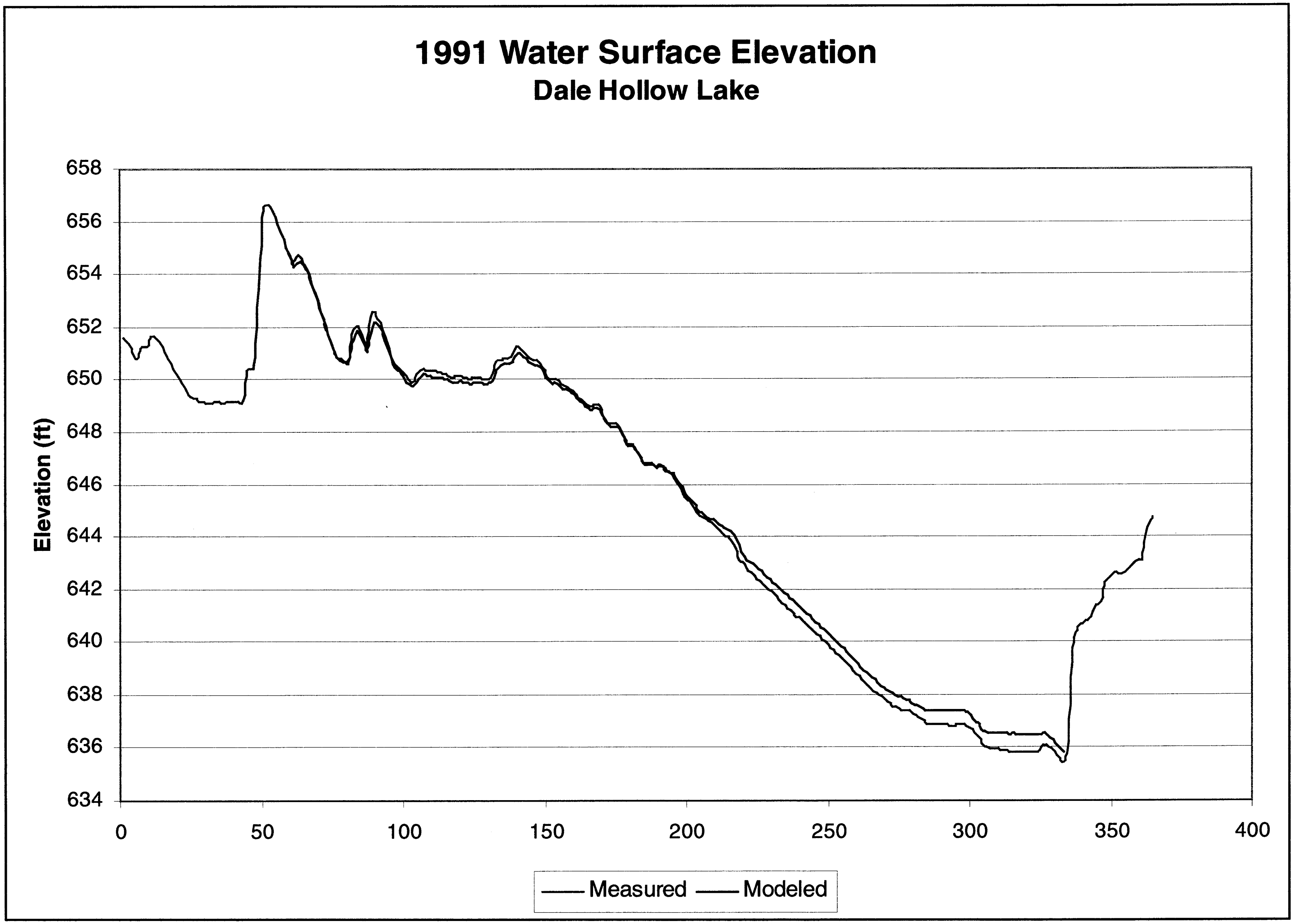


Figure 4.18. 1991 Measured and modeled Dale Hollow Lake pool elevation.

4.2.1 Input Data Assimilation and Synthesis

Required input data for a hydrodynamic and temperature simulation with the CE-QUAL-W2 model include:

1) Bathymetric and hydrologic inputs,
2) Initial lake temperature data,
3) Inflow temperatures during the simulation period,
4) Meteorologic data during the simulation period, and
5) Values of selected physical coefficients.

4.2.2 Initial Temperatures

Simulations for each year were started on March 1, during isothermal conditions. This was done due to the absence of observed temperature profile data during early spring. An isothermal condition indicates that the temperature is uniform throughout the water column, therefore, a single temperature value was used as the initial water temperature for all layers represented in the model.

The earliest available profile for 1973 was measured on April 3. This profile indicated a bottom temperature of approximately 6°C, with a measured tailwater temperature of 8°C. Based on these data and subsequent model runs using varying initial temperatures, an initial temperature of 6°C was chosen for 1973. As mentioned previously, there were no early profiles for 1988 or 1991 from which to garner an initial temperature. Therefore iterative model runs yielded initial temperatures for 1988 and 1991 simulations of 7°C and 9°C, respectively.

4.2.3 Inflow Temperatures

Inflow temperature is important for simulating the proper vertical placement of inflows in the reservoir. Although any time interval may be used for specifying inflow temperatures, the use of daily observations is recommended. Daily observed water temperatures were not available for all of the Dale Hollow Lake inflows included in the model. Water temperatures were measured on East Fork Obey River, West Fork Obey River, Wolf River, Big Eagle Creek, and Spring Creek. Ultimately, the Big Eagle Creek and Spring Creek measured data were not used based on the small number of measurements. The Wolf River water temperatures were used to represent water temperatures for Big Eagle Creek, Spring Creek, Ashburn Creek, Sulphur Creek, Irons Creek, Holly Creek, Mitchell Creek, Illwill Creek, and the distributed tributaries. Wolf River was used because it was the minor tributary with the most measured temperature data and temperature exhibits the qualities of a small stream that were presumed similar to the other tributaries.

Because stream temperatures fluctuate in response to meteorological forces it was possible to estimate daily inflow temperatures based on variations in both the measured water temperatures and the daily air temperatures using regression analyses. The first step was to estimate the temperature cycles for the year. This was accomplished by fitting a sine curve to the observed temperature data for each parameter. Examples are given on Figures 4.19 and 4.20, which show the sine curves representing the temperature cycle estimated from the observed data for air temperatures at Livingston, TN (Figure 2.1) and for water temperatures in the East Fork Obey River. There were instances where air temperature measurements were missing from the Livingston station during our study years; i.e., the entire month of May 1973 was missing. In order to supplement this data the mean monthly air temperature of the next four closest stations were compared. The average daily May 1973 air temperatures were then taken from the station whose April and June 1973 mean air temperature were most like those recorded at the Livingston station for the same period. This station turned out to be Monticello in Wayne County, KY. In two other cases a single daily air temperature was missing. These temperature values were calculated by interpolation of the day previous and day following.

As mentioned above, the water temperature values used were measured at East Fork Obey River, West Fork Obey River, and Wolf River. For each day on which there was an observed value, a residual was calculated as the difference between the observed value and the seasonal cycle value. Then a linear regression analysis was performed using the observed water temperature residual as the dependent variable and the air temperature residuals and daily streamflow rates as independent variables. Because of its greater heat capacity, water temperature responds more slowly than air temperature to changes in meteorological conditions; therefore, the regression analysis included air temperature residuals from preceding days. The period of record used for these calculations was 1964 through 1991. Streamflow data was only

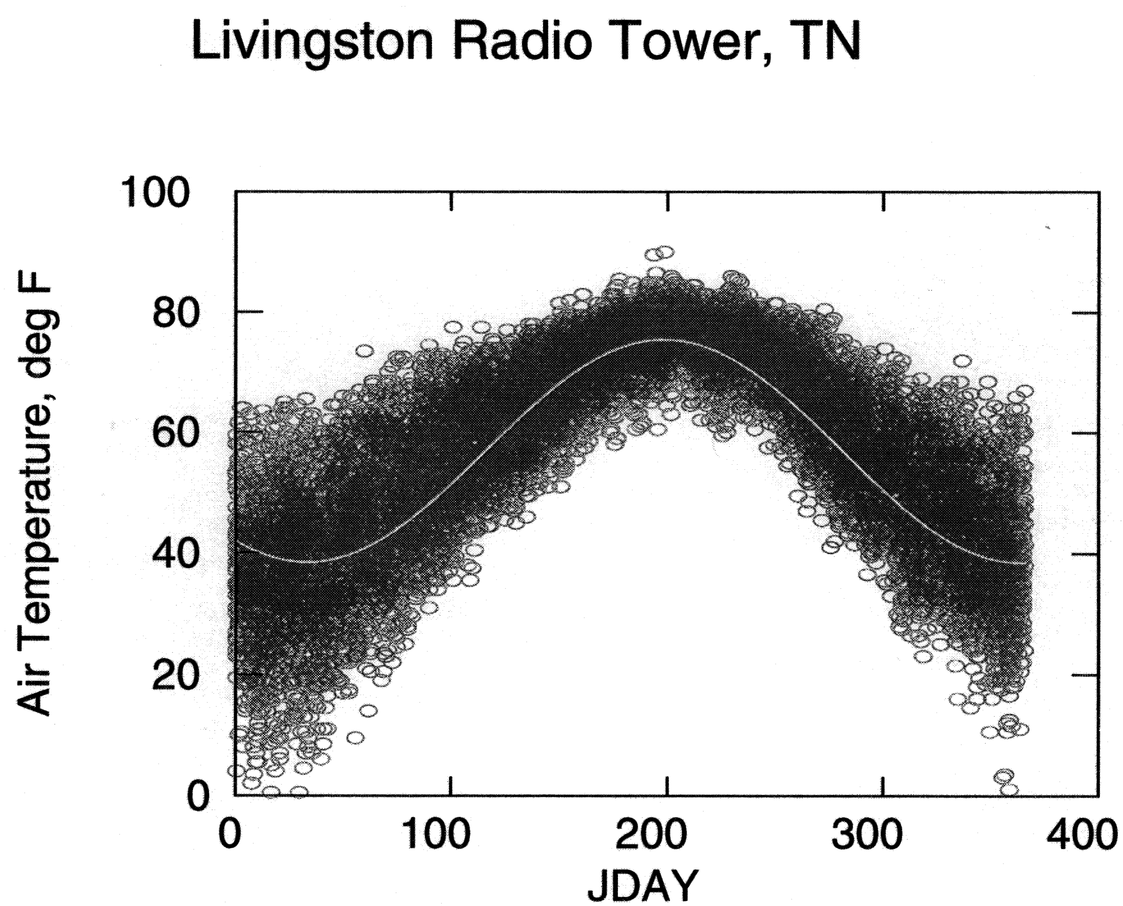


Figure 4.19. Seasonal air temperature curve of Livingston, TN.

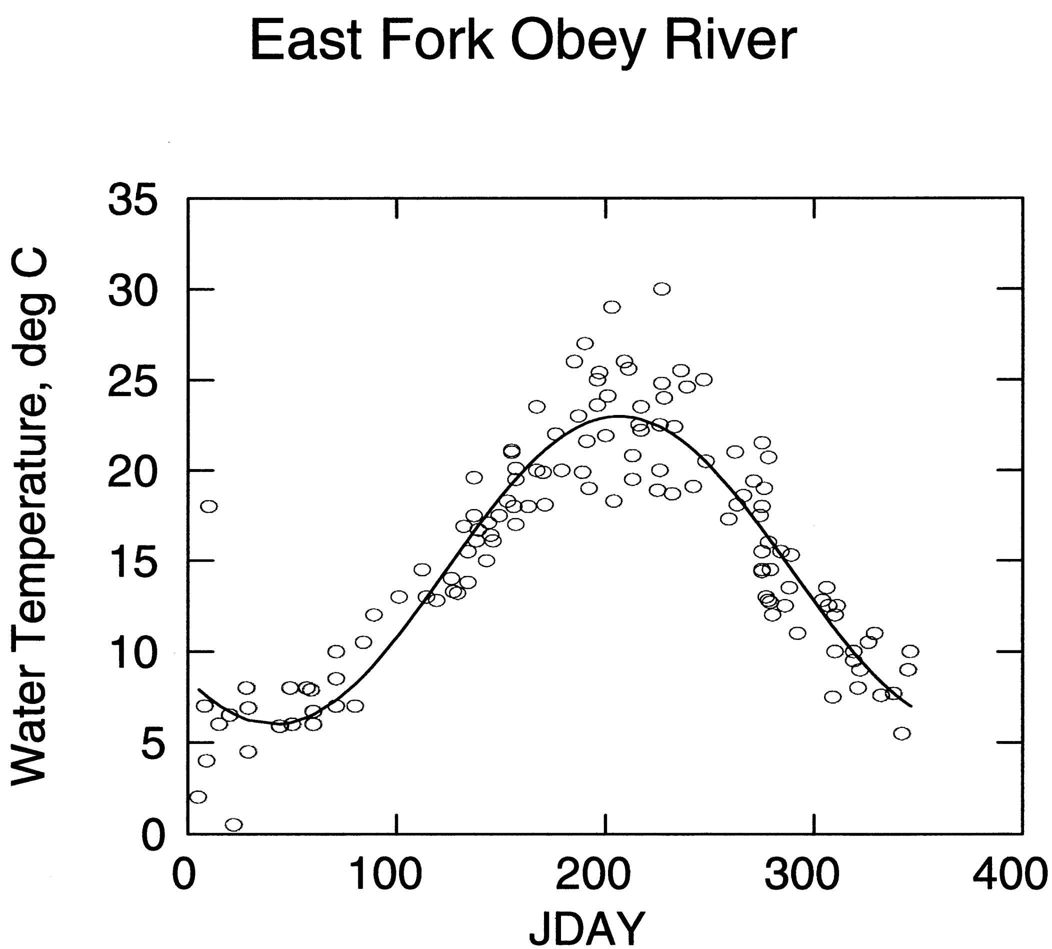


Figure 4.20. Seasonal water temperature curve for East Fork Obey River.

partially available and ultimately was excluded from the final regression equation. The equation developed for estimated water temperature for the day is shown below:

$$WT(t) = T(t) + a^* ATR(t) + b^* ATR(t\text{-}1) + c^* ATR(t\text{-}2) + d$$

where:
- WT(t) = water temperature for day t (°C)
- T(t) = water temperature for day t (°C) for sine curve
- ATR(t) = air temperature residual for day t (°C)
- ATR(t-1) = air temperature residual for day t-1 (°C)
- ATR(t-2) = air temperature residual for day t-2 (°C)
- a, b, c, d = coefficients

The daily water temperature residual equations developed for each stream are included in Table 4.3.

Table 4.3. Water temperature estimates.

Stream	Station	Equations	R^2
East Fork Obey River	3DAL10014 03414500	T(t) = 14.5 – 8.487* sine (0.019 * day + 0.793) WT(t) = T(t) – 0.216 + 0.063 * ATR(t) – 0.002 * ATR(t-1) + 0.082 * ATR(t-2)	R^2=0.84 R^2=0.15
West Fork Obey River	3DAL10015 03415000	T(t) = 15 – 8.807* sine (0.019 * day + 0.838) WT(t) = T(t) – 0.563 + 0.175 * ATR(t) + 0.063 * ATR(t-1) + 0.119 * ATR(t-2)	R^2=0.73 R^2=0.49
Wolf River	3DAL10011 03416000	T(t) = 15 – 9.348* sine (0.019 * day + 0.836) WT(t) = T(t) – 0.884 + 0.073 * ATR(t) + 0.118 * ATR(t-1) + 0.069 * ATR(t-2)	R^2=0.84 R^2=0.29

Daily inflow temperatures estimated from this procedure are included in Appendix H. These plots include the calculated and observed water temperatures for East Fork Obey River, West Fork Obey River, and Wolf River. The resulting estimated inflow water temperatures appeared to indicate the results for a specific year at a specific stream were not representative of the observed data. Specifically, it appeared that the 1988 estimated water temperatures on Wolf River were too cool. To validate this temperature estimation procedure, a sensitivity water temperature calibration simulation was done increasing these temperatures 2°C throughout the entire year. The result indicated no improvement in the temperature profile; therefore the water temperatures calculated by the equations above were maintained.

4.2.4 Meteorologic Data

The meteorologic input data used by CE-QUAL-W2 consists of air temperature, dew point temperature, wind speed, wind direction, and cloud cover. The meteorologic data utilized in the model were obtained from the Nashville International Airport station. Hourly data for the Nashville station for 1973, 1988, and 1991 were obtained from a CD-ROM (EarthInfo 1996) that contains data retrieved from the National Oceanic and Atmospheric Administration (NOAA).

The air temperatures and dew point temperatures recorded at the Nashville station are warmer than those at Livingston Radio Tower, the nearest station to Dale Hollow Lake. The Livingston station is located about 45 miles south of the center of the reservoir. Nashville is a relatively large metropolitan area, and it is likely that Nashville air temperatures are warmer than the surrounding countryside primarily due to the factors specific to a large city. Based on previous experience during calibration, the model predicts water surface temperatures that are too warm, when using the input of these higher "city" temperatures. During the study years, the average annual air temperature available for Livingston ranges from 0.7°C to 1.5°C cooler than Nashville during these same years. Therefore, the hourly input air temperatures were adjusted to reflect these cooler temperatures indicative of the area surrounding the reservoir. The adjustment methodology included first comparing the daily temperatures recorded at Livingston with those recorded at Nashville for the study years. The Livingston station included only daily temperature readings. The difference in these daily values was computed and applied to each hour of each respective day of the Nashville hourly temperatures. In essence, the Nashville temperatures were adjusted to reflect the temperatures recorded at Livingston. The dew point temperatures are also required for the model calibration and are not available at Livingston. In order to maintain the relationship between dew point and temperature, the relative humidity was calculated using Standard Psychrometric Tables. The revised air temperature and computed relative humidity were then used to compute a revised dew point temperature that maintained the relationship of the recorded Nashville air temperatures and dew point temperatures. The other components of the meteorologic data; wind speed, wind direction and cloud cover, were not recorded at Livingston, but at Nashville only, and were included without any modification.

4.2.5 Physical Coefficients

The CE-QUAL-W2 coefficients used to calibrate the temperature and hydrodynamic algorithms are shown in Table 4.4. The horizontal dispersion coefficients, AX and DX are for momentum and temperature/constituents, respectively. The CHEZY coefficient is used in calculating boundary friction. The values used for AX, DX, and CHEZY were model default values and were not changed during calibration. AX and DX are the horizontal dispersion coefficients for momentum, and temperature and other constituents, respectively. They are presently time and space invariant in the model.

Table 4.4. Physical coefficients used in hydrodynamic/temperature calibrations.

Mnemonic Name	Description	Range of Values Simulated	Final Value
AX	Longitudinal eddy viscosity	1.0 m^2/sec	1.0 m^2/sec
DX	Longitudinal eddy diffusivity	1.0 m^2/sec	1.0 m^2/sec
CHEZY	Chezy coefficient	70 $m^{0.5}$/sec	70 $m^{0.5}$/sec
WSC	Wind sheltering coefficient	0.40 – 1.00	0.55
WFC	Wind function coefficient	0.50 – 1.00	1.00
BETA	Fraction of solar radiation absorbed at surface	0.43	0.43
EXH20	Extinction coefficient for pure water	0.31 m^{-1}	0.31 m^{-1}
EXINOR	Extinction coefficient for inorganic solids	0.03 m^{-1}	0.03 m^{-1}
EXORG	Extinction coefficient for organic solids	0.09 m^{-1}	0.09 m^{-1}
CBHE	Coefficient of bottom heat exchange	3.5E – 6 m^2/sec 3.5E – 8 m^2/sec	3.5E – 7 m^2/sec
TSED	Sediment temperature 1973 1988 1991	 10 – 14.2°C 9 – 14.3°C 10 – 15.7°C	 10°C 9°C 10°C

4.2.5.1 Light Extinction Inputs

In the CE-QUAL-W2 model, the net light extinction consists of three components; these are extinction of light in pure water (EXH2O), extinction of light due to inorganic suspended solids (EXINOR), and extinction of light due to algae and detritus (EXORG). ALGLIT is used to convert algae to detritus; it does not affect light extinction.

The values for EXH2O and BETA were calculated using the following relationships from the user's manual:

EXH2O = 1.11 Zs (mean or max)-0.73
BETA = 0.27 ln (EXH2Omean) + 0.61

where Zs = Mean or the Max Secchi depth in meters.

Secchi depth measurements were recorded from 1971 to 1999 at several stations in Dale Hollow Lake and near the mouths of East Fork Obey River, West Fork Obey River, and the Wolf River. The secchi depth measurements were typically gathered once in the spring, once in the summer, and once in the fall, with occasional divergence from this pattern. The mean secchi depth ranged from 1.74 m to 4.48 m. The measurements recorded at each station were averaged to establish the mean value for each station. In addition to the mean secchi depth, the maximum secchi depth recorded for the period of record, at each station, was also extracted from the data. The EXH2O coefficient was calculated at each station using the mean secchi depth and the maximum secchi depth. The EXH2O coefficient using the average secchi depth was used to compute BETA. The EXH2O coefficient used in the model calibrations was obtained using the maximum secchi depth values at each station and then averaged to represent the Dale Hollow Lake system. The calculated EXH2O is 0.31 and BETA is 0.43. EXINOR and EXORG were chosen based on the values for EXH20 and BETA while maintaining the relationship to each other exhibited in the manual as default values. EXINOR and EXORG modify light extinction through the water column as a function of inorganic solids and organic solids, respectively. ALGLIT was maintained at the default value of 0.01.

4.2.5.2 Wind Sheltering Inputs

The wind sheltering coefficient (WSC) is a calibration parameter to account for differences between actual wind speed on the reservoir surface and measured data at the meteorological station. This coefficient accounts for factors such as sheltering of the reservoir by surrounding hills. The initial value of WSC was set at 1.0, which indicates no sheltering. Several simulations were made with different values of WSC to determine the sensitivity of the model and the WSC value that would provide the best representation of vertical mixing due to wind driven turbulence in all three calibration years. The most reasonable results were obtained using the value of 0.55 for WSC.

4.2.5.3 Wind Function Coefficient

The wind function coefficient applies a modifier to the wind sheltering coefficient. The initial or default value of 1.0 indicates no additional adjustment to the wind sheltering coefficient components will be calculated. This value was adjusted to 0.5 for sensitivity, however, the results indicated the WSC could be adjusted sufficiently for calibration. Therefore, WFC was maintained at the default setting of 1.0.

4.2.5.4 Bottom Heat Exchange Inputs

The initial value of the sediment temperature (TSED) was the average annual air temperature as recommended in the user's manual. The nearest station with historical air temperature data is the Livingston Radio Tower station. The daily air temperatures recorded at the Livingston station were averaged and computed for each of the three calibration years. These initial values resulted in bottom temperatures that were warmer than observed, so lower values were tried. The nearest match of observed and predicted bottom temperatures was when a TSED of 10°C was used for 1973 and 1991, and 9°C for 1988.

Another model coefficient that affects hypolimnetic temperatures is the coefficient of bottom heat exchange (CBHE). The initial value of CBHE was $3.5E^{-7}$ m^2/sec, which is the value used in a previous application of the model. After the sediment temperature was set, several sensitivity simulations were made with different values of CBHE to see which value would provide the best match between predicted and observed hypolimnetic temperatures in all three calibration years. The best results were achieved when using the original value for CBHE.

4.2.6 Outlet Specification

4.2.6.1 Outlet Centerline and Source Specification

The early simulation results of tailwater or outflow temperatures did not provide a good match with observed tailwater temperatures. This result was be partly attributed to differences between the predicted and observed temperature profiles in the pool. In order to improve the simulation, other issues with tailwater temperatures were investigated (e.g., specification of the outlet elevations in the control file, use of line source versus point source outflow, withdrawal zone computations, etc.). In an effort to raise the tailwater temperatures, the turbine outflow source was modified from a point sink outflow to line sink, with a width equal to the layer width at the centerline elevation. Although this modification did change results slightly, the outcome did not improve the tailwater temperatures. A second option was to adjust the centerline elevation of the turbine based on the temperatures indicated in the lake profiles. Verifying the withdrawal zone computed in CE-QUAL-W2 was comparable to the withdrawal zone computed with SELECT could validate this change of centerline elevation. Copy of the SELECT model output, which included a profile on the same day/year, was acquired and a profile compared from CE-QUAL-W2. The comparison verified that the withdrawal zones between the two models were of similar shape and included the same elevation range. In addition, the revised centerline elevation provided better output with respect to the observed tailwater temperatures. The final computed tailwater temperatures compared to measured data is included as Figure 4.21. The final turbine centerline elevation used in the simulation is 178.0 m, msl which is 3.3 m above the centerline elevation used in the SELECT model. (Refer to Appendix E)

4.3 Temperature Calibration Results

Observed and predicted temperature profiles are included in Appendix I. The model reproduced the overall shape of the temperature profiles and appeared to correctly predict the timing of the onset of stratification and fall overturn.

4.3.1 Temperature Profiles

During the 1973 wet year calibration, with respect to the shape of the temperature profile, the model predictions match fairly well. In general, the predicted surface and bottom temperatures are consistently within a few degrees of observed temperatures. The predicted surface temperatures tended to be slightly warmer while the predicted hypolimnetic temperatures tended to be slightly cooler than the observed temperatures.

The shape of the model-predicted temperature profile best fit the shape of the observed data for the dry year 1988. The predicted surface and bottom temperatures are consistently within a few degrees of observed temperatures, however, the bottom temperatures seem to become cooler as you move upstream through the lake. The metalimnion seems to be slightly warmer throughout the lake, however the shape of the observed profiles is mimicked with the predicted values.

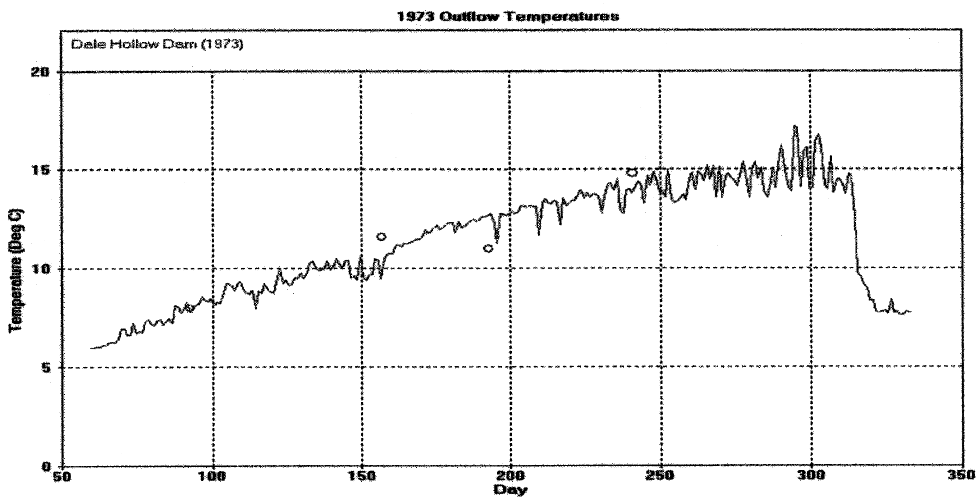
In the normal year, 1991, the predicted profiles are available only twice at any given station and then only later in the year. Even with this constraint the predicted temperature profiles generally match observed data well. However, the model tends to predict a slightly warmer and deeper metalimnion.

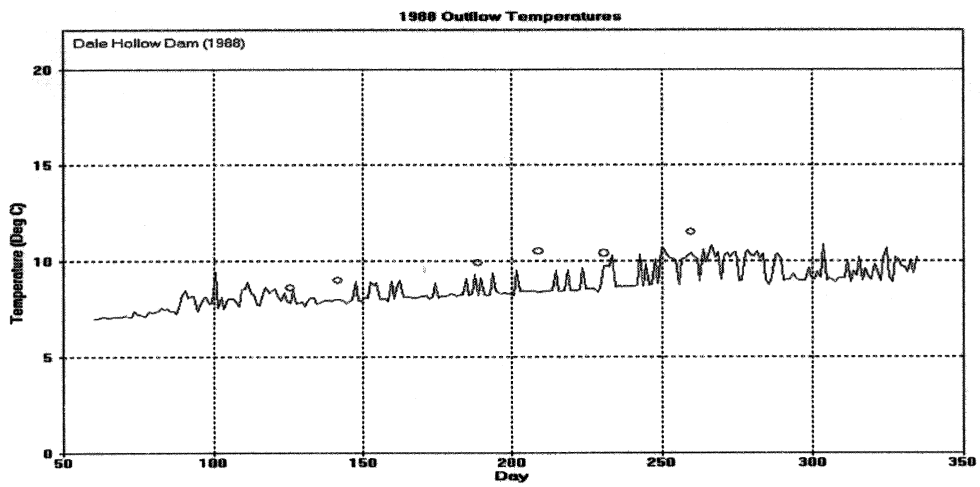
4.3.2 Surface Temperatures

Plots of observed and predicted surface temperature at the dam are shown on Figure 4.22. For all years, the model predicted temperatures within 1 to 3°C of observed data. During 1973 the predicted surface temperatures tended to be slightly warmer in the spring and early summer and then tended toward cooler in the late summer and fall. In all cases the predicted values were close to those measured. For 1988, the predicted temperatures match most of the observed data with the model, predicting slightly cooler temperatures upstream in the lake.

4.3.3 Outflow Temperatures

Predicted and observed outflow temperatures were plotted to ensure that the selective withdrawal algorithm in the model was properly configured (Section 4.2.6, Figure 4.21). The predicted outflow temperatures follow the general trends of the observed outflow temperatures. For the most part, predicted outflow temperatures are within 2 to 3°C of observed temperatures and typify the slopes indicated by observed data. Differences between predicted and observed outflow temperatures appear to be the result of differences between predicted and observed temperature profiles in the lake at the dam. Thus, predicted outflow temperatures are cooler than observed when the predicted temperature profile at the dam is cooler than the observed profile at the withdrawal elevation. Predicted outflow temperatures are warmer than observed when the





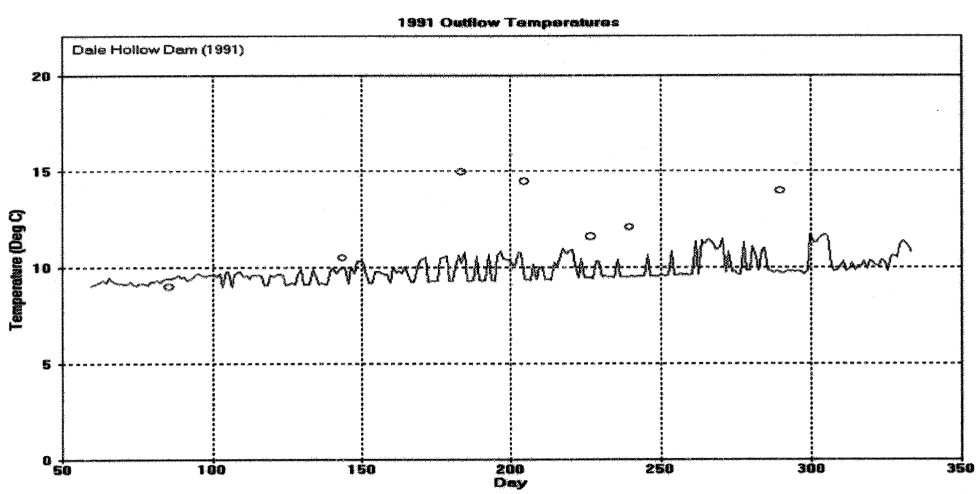


Figure 4.21. Comparison of measured and modeled release temperatures at Dale Hollow Dam.

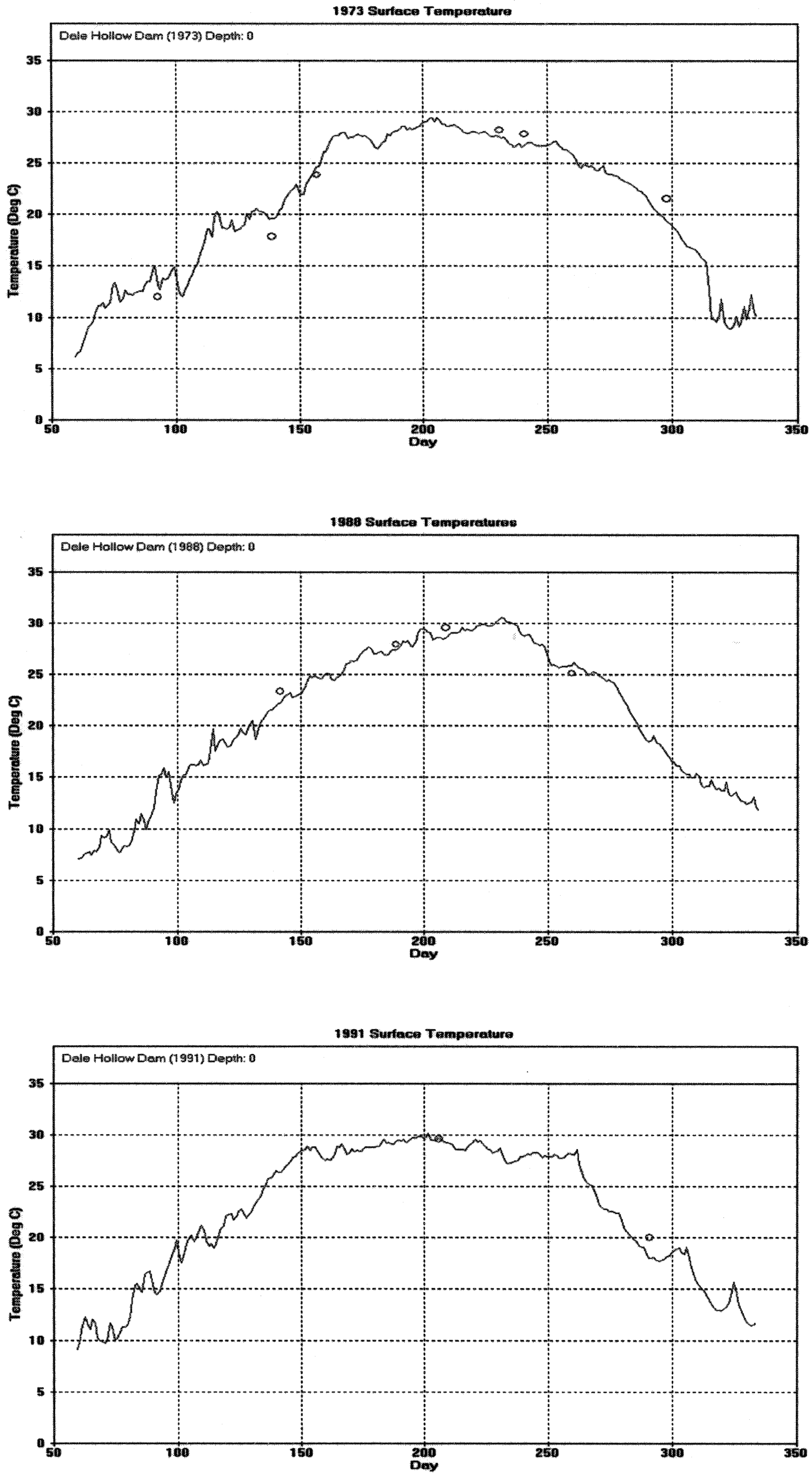


Figure 4.22. Comparison of measured and modeled surface temperatures at Dale Hollow Dam.

predicted temperature profile at the dam is warmer than the observed profile at the withdrawal elevation.

4.3.4 Conclusions

The calibrated temperature model reproduced the patterns observed in the lake as well as predicting temperatures, typically within 1 to 3°C of observed measurements. The surface temperatures and the outflow temperatures were equally well calibrated with only slight differences from year to year. The temperature calibration exercise has provided additional insight into the processes influencing inflow and reservoir water quality.

4.4 Water Quality Calibration

Water quality constituents of interest for this modeling effort were DO, nutrients, algae, and iron. The approach for calibrating reservoir water quality was to (1) prepare the input data required for the model; (2) compile the coefficient and parameter distributions based on existing water quality, and scientific literature for similar systems; and (3) use these coefficient and parameter distributions to initiate the calibration process for the 1973, 1988, and 1991 study years.

4.4.1 Input Constituents

Inflowing constituent loadings from eleven tributaries and direct runoff were required to simulate Dale Hollow Lake water quality. The tributaries were East Fork Obey River, West Fork Obey River, Wolf River, Spring Creek, Illwill Creek, Big Eagle Creek, Sulphur Creek, Holly Creek, Mitchell Creek, Ashburn Creek, and Irons Creek. Constituent inflow also entered the lake via the distributed tributaries. Measurements of DO, dissolved organic matter (labile and refractory DOM), silica, detritus, ammonia-N, nitrate+nitrite-N (hereafter referred to as nitrate-N), dissolved phosphorus/orthophosphorus, and iron were required for the reservoir inflows included in the model.

Temperature inputs were discussed in a previous section (Section 4.3). Water quality constituents were measured by the District, the Environmental Protection Agency (EPA), KYDNR, and the United States Geologic Survey (USGS) during the study years. The water quality data collected by the District included information at some/all stations during each of the study years for the lake and inflow streams. The water quality data obtained from the EPA included inlake and inflow streams for the years 1973 and 1974. These EPA data corresponded to an intensive National Eutrophication Study (NES) study published in 1978. During 1988, KYDNR collected water quality data in several arms of the lake lying on the Kentucky side of the reservoir. The water quality data from the USGS was not specific to the study years but was used only to develop historical mean sample values for iron and silica for inflow streams. Water quality data for the study years were obtained from one or more of the recording agencies discussed above for the following inflow streams: East Fork Obey River, West Fork Obey River, Wolf River, Spring Creek, Illwill Creek, and Big Eagle Creek. The data on Big Eagle included several years of data, but only one of the study years, 1973. In the compilation of water quality data, there were often constituent measurements recorded at or below the detection limit. In all cases the value of the detection limit was conservatively considered the measured value for the constituent. The detection levels of the various water quality parameter often varied depending on the agency managing the sample collections, the lab used in the processing of the water samples, and the year the samples were collected.

Water quality measurements were not collected during any of the study years in Ashburn Creek, Sulphur Creek, Irons Creek, Mitchell Creek, or Holly Creek. Water quality constituent inputs for these minor tributaries and the distributed tributary were typically set to values determined for Big Eagle Creek. The water quality data from Big Eagle Creek was used primarily because of the size of the drainage area and the similarity of its land use to that of the unmonitored creeks. The drainage area of Big Eagle Creek is 45 square miles, which was the smallest of the measured inflow streams. The unmeasured tributaries had drainage areas equal to or less than 20 square miles, with the exception of the distributed tributary, which included 260 square miles. However, the drainage area of the distributed tributary was "distributed" around the entire lake. The land use of the Big Eagle Creek watershed is primarily undeveloped mixed forestland directly surrounding the creek with cropland and pasture in the upper end of the watershed. Each of the minor tributaries also has this same general forested landuse characteristic with farmland in the outer reaches of the drainage area. In addition, Big Eagle

Creek is located on the southern side of the lake, adjacent to the majority of the minor tributaries, which further substantiates their similar land use and soil/vegetation classifications.

In 1973, measurements of dissolved orthophosphorus, ammonia-N, and nitrate-N were taken every month from April through November in each of the NES monitored tributary streams. These monitored streams included East Fork Obey River, West Fork Obey River, Wolf River, Spring Creek, Illwill Creek, and Big Eagle Creek. Daily input values were interpolated between the measured data. The typical starting value on day 59/60 for each of the tributaries was based on the initial concentration being used for the in-lake data or the initial measurement, assuming the value was similar to the initial concentration. In all cases the final measured data remained constant through the end of the model year. For the unmonitored tributaries, the Big Eagle Creek historical median was used for orthophosphorus, ammonia-N, and nitrate-N. The unmonitored tributaries include Ashburn Creek, Sulphur Creek, Irons Creek, Holly Creek, Mitchell Creek, and distributed tributaries.

In 1988 and 1991 there were no intensive water quality studies. Measured data was available for phosphorus, ammonia-N, and nitrate-N at the same monitored tributaries as discussed above with the exception of Big Eagle Creek. The number of measurements for 1988 and 1991 ranged from 1 to 4 sample dates. Estimated daily water quality concentrations were based on historical data from the tributary monitoring stations. Historical water quality data were evaluated to determine if relationships between water quality and flow or water temperature, or seasonal variability were evident. Initially, the data were examined by plotting the historical data from each monitoring station versus day of year, flow, and water temperature, untransformed and log transformed. In all cases where the measured value was "less than" detection level, the detection level value was included as the measured value. If linear or non-linear relationships were evident in the plots, they were tested for statistical significance using regression analysis. When no significant relationship could be determined for a parameter at a monitoring station, the model-input value was set to a constant value. Usually, this constant value was the historical median concentration. The historical median was then evaluated compared to the measured data that was available. Estimated values were examined for reasonableness and input values were modified to use the measured data whenever possible, while eliminating the influence of outliers.

Table 4.5 located at the end of this section defines the water quality tributary flow input values used and their relationship to the measured data or historical data for each of the study years. Although 1985 was not selected as a study year, a significant amount of tributary water quality sampling was done. A comparison of the 1985 data and the 1988 data was done to determine if a relationship might exist between these two years, both of which were very dry years. Unfortunately no such relationship was found; therefore the observed data used were limited to the small amount of tributary data available during 1988. Plots of the inflow concentration along with the measured concentration are included in Appendix J.

Daily DO values were important for modeling the reservoir. DO was not measured in any of the tributaries during 1973, and sparsely for 1988 and 1991. DO was calculated using the computed water temperatures. The equations associated with the computation of DO are also included in Table 4.5. The water temperatures for Dale Hollow Lake were computed for three stations; East Fork Obey River, West Fork Obey River, and the Wolf River. (The computation of water temperatures is discussed in Section 4.2.3.) The Wolf River water temperatures were used for all other tributaries of the reservoir. The percent saturation was calculated for each of the inflow tributaries and used in the computation for the inflow DO. There were inflow tributaries with percent saturation that was calculated to be in excess of 100%. In these cases, a max of 100% saturation was used. The inflow tributaries where the DO was considered 100% saturated include East and West Fork Obey River, Illwill Creek, and Spring Creek. The remaining inflow tributaries, including distributed tributaries, indicated a DO saturation of 99%. Plots of the inflow DO concentration along with the measured DO concentration are included in Appendix J.

The inflow constituent values for silica were obtained by taking the average of the available measurements on the inflow tributaries. The silica measurements were typically done by the USGS on the main tributaries, specifically East Fork Obey River, West Fork Obey River, and Wolf River. The values used during each of the study years were the historical mean, although the number of measurements did not exceed 4 on any tributary. East Fork Obey River yielded the highest average concentration of 7 mg/L, while both West Fork Obey River and Wolf River recorded an average of 5 mg/L. The silica concentration for the unmonitored tributaries is a loosely based function of drainage area size. The silica inflow concentration used for the unmonitored tributaries with drainage areas greater than 20 square miles was set to 2 mg/L. When the drainage area was equal to or less than 20 square miles on an unmonitored tributary, 1 mg/L of silica was used as the inflow concentration.

Iron was included for the Dale Hollow Lake study. Total iron measurements were recorded by the USGS and the District on several tributaries. The USGS measured iron on East Fork Obey River, West Fork Obey River, and Wolf River 22 to 23 times from 1965 to 1985. The District measured total iron on the previously listed tributaries plus Illwill Creek and Spring Creek. The 3 main tributaries were sampled only once by the District, however Illwill Creek and Spring Creek were sampled 4 and 21 times respectively. The historical average measurement for the particular tributary was used as a constant value in each of these inflow constituent files unless observed data were available to use. The iron concentration for the unmonitored tributaries was set equal to the mean total iron concentrations measured on Illwill Creek. This creek was chosen to represent the unmonitored tributaries because its watershed, as well as those it represents, did not appear to include areas with a mining land-use designation. A generalized land use map developed from National Land Cover Data (NLCD) was used to locate areas with mining land use designations (NLCD 2000). Although mining land use designations exist, all mines in the watershed are closed (US ACOE 1986). The unmonitored tributaries include Big Eagle Creek, Sulphur Creek, Ashburn Creek, Mitchell Creek, Irons Creek, Holly Creek, and distributed tributaries.

As discussed above, measurements of one or more of the inflow constituents of interest were typically available for 1973, 1988 and 1991. The exceptions were detritus and dissolved organic matter (DOM). Neither detritus nor DOM had been a monitored water quality constituent in the tributaries. Therefore, refractory DOM and detritus inflow inputs had to be estimated. Dissolved organic carbon and total organic carbon are typically used in the determination of detritus and DOM. However, there was not enough measurements in the tributaries any time during the period of record to develop a relationship. The alternate method used for estimating detritus incorporated measured total kjeldahl nitrogen (TKN) and ammonia-N, which were available. Particulate organic nitrogen for a stream was estimated by subtracting the measured ammonia from the measured TKN during the study years. Particulate organic carbon was estimated from this particulate organic nitrogen using the Redfield Ratio of C:N:P = 40:7:1 on a mass basis. This results in a C:N ratio of 5.7:1. Therefore, particulate organic carbon was estimated as 5.7 * particulate organic nitrogen. The particulate organic carbon was assumed to represent detritus. Refractory DOM was assumed to be 10% of detritus. Labile DOM was assumed to be negligible in inflows, so that constituent was set to zero in the inflow concentration files. During 1973, there were typically 8 measurements of TKN and ammonia-N as compared with 1 to 3 measurements for 1988 and 1991. In the estimation of detritus more data did not prove to be beneficial during 1973. The resulting estimated particulate organic carbon computed to represent detritus produced values that appeared to be errant numbers, such as values greater than 8, and as high as 25. Detritus values this high did not seem representative, even in a wet year. The range of estimated detritus for 1988 and 1991 (albeit less measured data) was between 0.5 and 2.0. In order to validate the 1973 detritus estimate, the values were compared with the measured flows (or calculated flow). Due to a relationship between flow and detritus, it was possible to evaluate which estimated values were erroneous. The detritus was plotted with flow on each of the respective tributaries for 1973. Values used in the model validated the relationship between flow and detritus. This method typically yielded 2 to 3 estimated values for each tributary that were then interpolated through the year. Plots of the estimated detritus values used for model inputs for 1973, 1988 and 1991are located in Appendix J.

Table 4.5. Water quality tributary inflow input methodology.

Tributary	Station	Water Quality Estimate
East Fork Obey River (OR)	3DAL10014 2102E1 034145000	Labile DOM = 0mg/L (1973, 1988, 1991) Refractory DOM = 10% of computed detritus. (1973, 1988, 1991) Silica = 7 mg/L (average of 3 measurements). (1973, 1988, 1991) Detritus = (measured TKN-measured NH_3)*5.7; values discarded when not correlated with flow. Initial measurement to start or estimated interpolation to next value, final value to end of model year. (1973) Detritus = 0.57 mg/L (measured TKN-measured NH_3)*5.7; single value. (1988) Detritus = (measured TKN-measured NH_3)*5.7; initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1991) Phosphorus/Orthophosphorus = Values interpolated from measured data; initial values used to start model year. (1973) Phosphorus = 0.01 mg/L (observed measurement). (1988) Phosphorus = start with first measurement, decreases to second measurement and return to value of first measurement. (1991) Ammonia-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.05 mg/L. (1973) Ammonia-N = 0.1 mg/L (appears to be detection limit). (1988) Ammonia-N = 0.1 mg/L (observed measurement & appears to be detection limit). (1991) Nitrate-N = Values interpolated from measured data; initial values used to start model year. (1973) Nitrate-N = initial value is first measurement, interpolated to 1/2 second measurement, decrease to median of 0.22 mg/L. (1988) Nitrate-N = initial value is median of 0.22 mg/L; decrease to observed data. (1991) DO ={ [exp $^{(7.7117 - 1.314*\ln(\text{temperature, °C} + 45.93))}$] * $[1-(H \div 44.3)]^{5.25}$ }* %Saturation/100 (See explanation of terms below). (1973, 1988, 1991) Iron (FE) = 1.23 mg/L (historical average). (1973, 1988) Iron (FE) = 0.129 mg/L(observed measurement) (1991)
West Fork Obey River (WO)	3DAL10015 2102H1 03415000	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = 10% of computed detritus. (1973, 1988, 1991) Silica = 5 mg/L (average of 3 measurements). (1973, 1988, 1991) Detritus = (measured TKN-measured NH_3)*5.7; values discarded when not correlated with flow. Initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1973) Detritus = 1.24 mg/L (based on comparative drainage area tributary with measured data). (1988)

Table 4.5 Continued

Tributary	Station	Water Quality Estimate
		Detritus = (measured TKN-measured NH_3)*5.7; initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1991) Phosphorus/Orthophosphorus = Values interpolated from measured data; initial values used to start model year. (1973) Phosphorus = 0.01 mg/L (historical median). (1988) Phosphorus = 0.01 mg/L (observed data & historical median). (1991) Ammonia-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.05 mg/L. (1973) Ammonia-N = initial value is median of 0.08 mg/L, increases to measured data and held to end of model year. (1988) Ammonia-N = initial value median of 0.08 mg/L, increase to first & second measured values, decrease back to median and hold. (1991) Nitrate-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.22 mg/L. (1973) Nitrate-N = 0.26 mg/L (historical median). (1988) Nitrate-N = initial value is median of 0.26 mg/L, increases to first measured data, decreases to second value and decreases to median and held to end of model year. (1991) DO ={ [exp $^{(7.7117 - 1.314 * \ln (\text{temperature, °C} + 45.93))}$] * $[1-(H \div 44.3)]^{5.25}$ }* %Saturation/100 (See explanation of terms below). (1973, 1988, 1991) Iron (FE) = 0.61 mg/L (historical average). (1973) Iron (FE) = 0.1 mg/L (based on observed measurement). (1988, 1991)
Wolf River (WR)	3DAL10011 2102D1 03416000	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = 10% of computed detritus. (1973, 1988, 1991) Silica = 5 mg/L (average of 4 measurements). (1973, 1988, 1991) Detritus = (measured TKN-measured NH_3)*5.7; values discarded when not correlated with flow. Initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1973) Detritus = 1.14 mg/L (measured TKN-measured NH_3)*5.7; single value. (1988) Detritus = (measured TKN-measured NH_3)*5.7; initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1991) Phosphorus/Orthophosphorus = Values interpolated from measured data; start of model year with initial inflow concentration of 0.01mg/L. (1973) Phosphorus = 0.01 mg/L (observed value). (1988) Phosphorus = initial value is median of 0.015 mg/L and decrease to observed data and hold. (1991)

Table 4.5 Continued

Tributary	Station	Water Quality Estimate
		Ammonia-N = Values interpolated from measured data; start of model year with initial inflow concentration of 0.05 mg/L. (1973) Ammonia-N = initial value is first measurement, increased to second measurement, decrease to first measurement. (1988) Ammonia-N = 0.1 mg/L (observed data & historical median). (1991) Nitrate-N = Values interpolated from measured data; initial values used to start model year. (1973) Nitrate-N = initial value is median of 0.36mg/L, increase to first and second measurement, drop back to median. (1988) Nitrate-N = 0.1 mg/L (observed data). (1991) DO ={ [exp $^{(7.7117 - 1.314*\ln(\text{temperature, °C} + 45.93))}$] * [1-(H÷44.3)]$^{5.25}$ }* %Saturation/100 (See explanation of terms below). (1973, 1988, 1991) Iron (FE) = 0.36 mg/L (historical average). (1973, 1988) Iron (FE) = 0.14 mg/L (based on observed measurement). (1991)
Illwill Creek (IL)	3DAL10016 2102C1 CLN116 CLN117	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = 10% of computed detritus. (1973, 1988, 1991) Silica = 2 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1973, 1988, 1999) Detritus = (measured TKN-measured NH_3)*5.7; values discarded when not correlated with flow. Initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1973) Detritus = 0.57 mg/L (measured TKN-measured NH_3)*5.7; initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1988, 1991) Phosphorus/Orthophosphorus = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.01mg/L. (1973) Phosphorus = 0.01 mg/L (observed value & historic median). (1988) Phosphorus = initial value is median of 0.01mg/L, increase to first measurement, decrease to second and hold. (1991) Ammonia-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.05 mg/L. (1973) Ammonia-N = 0.1 mg/L (observed value). (1988) Ammonia-N = 0.1 mg/L (observed data & detection limit). (1991) Nitrate-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.22 mg/L. (1973) Nitrate-N = initial value is first measurement, increased to second measurement, decrease to first measurement. (1988) Nitrate-N = initial value is median of 0.4 mg/L; decrease to first then second measurement, hold to end of model year. (1991)

Table 4.5 Continued

Tributary	Station	Water Quality Estimate
		DO ={ [exp $^{(7.7117 - 1.314*\ln(\text{temperature, °C} + 45.93))}$] * [1-(H÷44.3)]$^{5.25}$ }* %Saturation/100 (See explanation of terms below). (1973, 1988, 1991) Iron (FE) = 0.38 mg/L (historical average). (1973, 1988) Iron (FE) = 0.17 mg/L (based on observed measurement). (1991)
Big Eagle Creek (BE)	3DAL10028 2102J1	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = 10% of computed detritus. (1973, 1988, 1991) Silica = 2 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1973, 1988, 1991) Detritus = (measured TKN-measured NH_3)*5.7; values discarded when not correlated with flow. Initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1973) Detritus = 0.48 mg/L (based on comparative drainage area size to monitored tributaries). (1988) Detritus = 1.077 mg/L (based on comparative drainage area of tributary with measured data). (1991) Phosphorus/Orthophosphorus = Values interpolated from measured data; initial values used to start model year. (1973) Phosphorus = 0.01 mg/L (historical median). (1988) Phosphorus = 0.01 mg/L. (1991) Ammonia-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.05 mg/L. (1973) Ammonia-N = 0.1 mg/L (detection limit, no measured values). (1988) Ammonia-N = 0.1 mg/L (detection limit of OR, no measured data). (1991) Nitrate-N = Values interpolated from measured data; initial values used to start model year. (1973) Nitrate-N = 0.23 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1988) Nitrate-N = 0.1 mg/L (detection limit of OR no measured data). (1991) DO ={ [exp $^{(7.7117 - 1.314*\ln(\text{temperature, °C} + 45.93))}$] * [1-(H÷44.3)]$^{5.25}$ }* %Saturation/100 (See explanation of terms below). (1973, 1988, 1991) Iron (FE) = 0.38 mg/L (based on monitored drainage area without mines). (1973, 1988) Iron (FE) = 0.17 mg/L (based on monitored drainage area without mines). (1991)

Table 4.5 Continued

Tributary	Station	Water Quality Estimate
Spring Creek (SP)	3DAL10012 3DAL10013 2102A1 CLN119	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = 10% of computed detritus. (1973, 1988, 1991) Silica = 2 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1973, 1988, 1991) Detritus = (measured TKN-measured NH_3)*5.7; values discarded when not correlated with flow. Initial value extended to start or estimated, interpolation to next value, final value to end of model year. (1973) Detritus = (measured TKN-measured NH_3)*5.7; initial measurement to start or estimated, interpolation to next value, final value to end of model year. (1988, 1991) Phosphorus/Orthophosphorus = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.01 mg/L. (1973) Phosphorus = initial value is median of 0.037 mg/L, rise to first measurement and drop back down to median. (1988) Phosphorus = initial value is median of 0.037 mg/L, decrease to first measurement, increase to second, decrease to median and hold. (1991) Ammonia-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.05 mg/L. (1973, 1988, 1991) Nitrate-N = Values interpolated from measured data; start of model year with initial in-lake concentration of 0.22 mg/L. (1973, 1988, 1991) DO ={ [exp $^{(7.7117 - 1.314*\ln(\text{temperature, °C} + 45.93))}$] * [1-(H÷44.3)]$^{5.25}$ }* %Saturation/100 (See explanation of terms below). (1973, 1988, 1991) Iron (FE) = 0.26 mg/L (historical average). (1973, 1988) Iron (FE) = 0.10 mg/L (observed measurement). (1991)
Ashburn Creek (AC), Irons Creek (IC), Mitchell Creek (MC)	3DAL10027 (IC)	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = copied from Big Eagle Creek. (1973, 1988, 1991) Silica = 1 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1973, 1988, 1991) Detritus = copied from Big Eagle Creek. (1973) Detritus = 0.22 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1988) Detritus = 0.487 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1991) Orthophosphorus = 0.008 mg/L; Big Eagle Creek historical median. (1973) Phosphorus = 0.01 mg/L; Big Eagle Creek historical median (1988, 1991) Ammonia-N = 0.047 mg/L; Big Eagle Creek historical median. (1973)

Table 4.5 Continued

Tributary	Station	Water Quality Estimate
		Ammonia-N = 0.1 mg/L; detection limit on OR; as used on Big Eagle Creek. (1988, 1991) Nitrate-N = 0.53 mg/L; Big Eagle Creek historical median. (1973) Nitrate-N = 0.23 mg/L; see Big Eagle Creek. (1988) Nitrate-N = 0.10 mg/L; see Big Eagle Creek (1991) DO = copied from Big Eagle Creek. (1973, 1988, 1991) Iron (FE) = 0.38 mg/L (based on monitored drainage area without mines). (1973, 1988) Iron (FE) = 0.17 mg/L (based on monitored drainage area without mines). (1991)
Sulphur Creek (SC)	3DAL10031	Labile DOM = 0 mg/L (1973, 1988, 1991) Refractory DOM = copied from Big Eagle Creek. (1973, 1988, 1991) Silica = 1 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1973, 1988, 1991) Detritus = copied from Big Eagle Creek. (1973) Detritus = 0.27 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1988) Detritus = 0.616 mg/L (estimated based on comparative draiage area size to monitored tributaries). (1991) Orthophosphorus = 0.008 mg/L; Big Eagle Creek historical median. (1973) Phosphorus = 0.01 mg/L; Big Eagle Creek historical median. (1988, 1991) Ammonia-N = 0.047 mg/L; Big Eagle Creek historical median. (1973) Ammonia-N = 0.1 mg/L; detection limit on OR; as used on Big Eagle Creek. (1988, 1991) Nitrate-N = 0.53 mg/L; Big Eagle Creek historical median. (1973) Nitrate-N = 0.23 mg/L; see Big Eagle Creek. (1988) Nitrate-N = 0.10 mg/L; see Big Eagle Creek. (1991) DO = copied from Big Eagle Creek. (1973, 1988, 1991) Iron (FE) = 0.38 mg/L (based on monitored drainage area without mines). (1973, 1988) Iron (FE) = 0.17 mg/L (based on monitored drainage area without mines). (1991)
Holly Creek (HC) & Distributed Tributaries (DT)	None	Labile DOM = 0 (1973, 1988, 1991) Refractory DOM = copied from Big Eagle Creek. (1973, 1988, 1991) Silica = 1 mg/L (estimated based on comparative drainage area size to monitored tributaries). (1973, 1988, 1991)

Table 4.5 Continued

Tributary	Station	Water Quality Estimate
		Detritus = copied from Big Eagle Creek. (1973) Detritus = 0.11 mg/L for HC; 2.80 mg/L for DT, (estimated based on comparative drainage area size to monitored tributaries). (1988) Detritus = 0.231 mg/L for HC, 6.284 mg/L for DT (estimated based on comparative drainage area size to monitored tributaries). (1991) Orthophosphorus = 0.008 mg/L; Big Eagle Creek historical median. (1973) Phosphorus = 0.01 mg/L; Big Eagle Creek historical median. (1988, 1991) Ammonia-N = 0.047 mg/L; Big Eagle Creek historical median. (1973) Ammonia-N = 0.1 mg/L; detection limit on OR; as used on Big Eagle Creek. (1988, 1991) Nitrate-N = 0.53 mg/L; Big Eagle Creek historical median. (1973) Nitrate-N = 0.23 mg/L; see Big Eagle Creek. (1988) Nitrate-N = 0.10 mg/L; see Big Eagle Creek. (1991) DO = copied from Big Eagle Creek. (1973, 1988, 1991) Iron (FE) = 0.38 mg/L (based on monitored drainage area without mines). (1973, 1988) Iron (FE) = 0.17 mg/L (based on monitored drainage area without mines). (1991)

Note: DO equation includes temperature °C which refers to the calculated daily water temperature. The water temperatures were calculated for OR, WO, and WR only. All other tributary water temperatures use the values calculated for WO. The H value is the normal pool elevation in kilometers, which is 0.198 km for Dale Hollow Lake. The % saturation value is 100% for OR, WO, IL, and SP; all others are 99%.

4.4.2 Model Coefficients

This section discusses the water quality coefficients, parameters, and constants. These water quality coefficients can be categorized as:

- Biological Coefficients (e.g., algal growth rates);
- Chemical Coefficients (e.g., nitrification rates, SOD, etc.);
- Rate Modifiers (e.g., temperature factors, Q10 factors, etc.); and
- Stoichiometric Constants (e.g., oxygen required to oxidize 1 mole of NH_3 to 1 mole of NO_3).

Coefficient values were compiled from the water quality literature for reservoirs with similar hydrologic, physical, water quality, and biological characteristics as Dale Hollow Lake. The initial coefficient values used to initiate model calibration were taken from these coefficient compilations. If necessary, this initial coefficient rate was modified to reduce the deviation between observed and predicted values during calibration. The coefficients, rate modifying parameters, and stoichiometric constants for the water quality variables simulated in Dale Hollow Lake are listed in Table 4.6 and discussed below.

Table 4.6. CE-QUAL-W2 parameters and coefficients.

Name	Description	Initial Value	Modified Value
AGROW	Algal growth rate, day^{-1}	1.20, 0.95, 1.00*	2.064, 2.05, 1.60*
AMORT	Algal mortality rate, day^{-1}	0.03, 0.03, 0.03*	0.03, 0.03, 0.03*
AEXCR	Algal excretion rate, day^{-1}	0.0, 0.0, 0.0*	0.0, 0.0, 0.0*
ARESP	Algal dark respiration rate, day^{-1}	0.07, 0.05, 0.04*	0.05, 0.05, 0.005*
ASETL	Algal settling rate, day^{-1}	0.35, 0.15, 0.08*	0.35, 0.15, 0.08*
ASATUR	Saturation intensity at maximum photosynthetic rate, Wm^{-2}	125, 120, 135*	86, 75, 85*
ABIOP	Algal stoichiometry, phosphorus	0.004, 0.004, 0.004*	0.004, 0.004, 0.004*
ABION	Algal stoichiometry, ammonia	0.067, 0.067, 0.067*	0.067, 0.067, 0.067*
ABIOSI	Algal stoichiometry, silica	0.0, 0.0, 0.0*	0.0, 0.0, 0.0*
ABIOC	Algal stoichiometry, carbon	0.5, 0.5, 0.5*	0.5, 0.5, 0.5*
ALGT1	Lower temperature for algal growth, °C	5, 5, 10*	8, 10, 12*
ALGT2	Lower temperature for maximum algal growth, °C	12, 10, 25*	14, 16, 25*
ALGT3	Upper temperature for maximum algal growth, °C	22, 25, 38*	20, 25, 38*
ALGT4	Upper temperature for algal growth, °C	35, 30, 42*	25, 30, 42*
AGK1	Fraction of algal growth rate at ALGT1	0.1, 0.1, 0.1*	0.1, 0.1, 0.1*
AGK2	Fraction of maximum algal growth rate at AGLT2	0.98, 0.98, 0.98*	0.98, 0.98, 0.98*
AGK3	Fraction of maximum algal growth rate at ALGT3	0.98, 0.98, 0.98*	0.98, 0.98, 0.98*
AGK4	Fraction of algal growth rate at ALGT4	0.1, 0.1, 0.1*	0.1, 0.1, 0.1, 0.1*
LABDK	Labile DOM decay rate, day^{-1}	0.3	0.3
LRFDK	Labile to refractory decay rate, day^{-1}	0.001	0.001
REFDK	Maximum refractory decay rate, day^{-1}	0.001	0.001
DETDK	Detritus decay rate, day^{-1}	0.06	0.006
DSETL	Detritus settling rate, day^{-1}	0.05	0.10
OMT1	Lower temperature for organic matter decay, °C	4	4
OMT2	Lower temperature for maximum organic matter decay, °C	20	20
OMK1	Fraction of organic matter decay rate at OMT1	0.1	0.1
OMK2	Fraction of organic matter decay rate at OMT2	0.98	0.98
SEDDK	Sediment decay rate, day^{-1}	0.06	0.06
SOD	Sediment oxygen demand (SOD) for each segment, $gm^{-2}\ day^{-1}$	0.75 to 3.0	0.5
PO4REL	Sediment release rate (fraction of SOD)	0.007	0.002
PARTP	Phosphorus partitioning coefficient for suspended solids	0.1	0.1

Table 4.6 Continued

Name	Description	Initial Value	Modified Value
AHSP	Algal half-saturation constant for phosphorus, *gm*$^{-3}$	0.012, 0.010, 0.015*	0.002, 0.003, 0.004*
NH3REL	Sediment release rate of ammonia (fraction of SOD)	0.035	0.035
NH3DK	Ammonia decay rate, *day*$^{-1}$	0.25	0.05
PARTN	Ammonia partitioning coefficient for suspended solids	0.01	0.01
AHSN	Algal half-saturation constant for ammonia, *gm*$^{-3}$	0.06, 0.06, 0.10*	0.06, 0.06, 0.10*
AHSSI	Algal half-saturation constant for silica, *gm*$^{-3}$	0.1, 0.0, 0.0*	0.1, 0.0, 0.0*
NH3T1	Lower temperature for ammonia decay, *°C*	0.1	0.1
NH3T2	Lower temperature for maximum ammonia decay, *°C*	20	20
NH3K1	Fraction of nitrification rate at NH3T1	0.1	0.1
NH3K2	Fraction of nitrification rate at NH3T2	0.98	0.98
NO3DK	Nitrate decay rate, *day*$^{-1}$	0.1	0.1
NO3T1	Lower temperature for nitrate decay, *°C*	5.0	5.0
NO3T2	Lower temperature for maximum nitrate decay, *°C*	20.0	20.0
NO3K1	Fraction of denitrification rate at NO3T1	0.1	0.1
NO3K2	Fraction of denitrification rate at NO3T2	0.98	0.98
SIREL	Silica release rate (fraction of sediment oxygen demand)	0.01	0.01
CO2REL	Sediment carbon dioxide release rate (fraction of sediment oxygen demand)	0.1	0.1
FEREL	Iron sediment release rate (fraction of sediment oxygen demand) (values of 0.3-0.5 used in past)	0.4	0.4
FESETL	Iron settling rate, *day*$^{-1}$ (values of 0.5-2.0 used in past)	0.5	0.5
O2NH3	Oxygen stoichiometric equivalent for ammonia decay	3.43	3.43
O2ORG	Oxygen stoichiometric equivalent for decay of organic matter	1.4	1.4
O2RESP	Oxygen stoichiometric equivalent for dark respiration	1.2	1.2
O2ALG	Oxygen stoichiometric equivalent for algal growth	1.4	1.4
BIOP	Stoichiometric equivalent between organic matter and phosphorus	0.004	0.004
BION	Stoichiometric equivalent between organic matter and nitrogen	0.067	0.067
BIOSI	Stoichiometric equivalent between organic matter and silica	0.0	0.0
BIOC	Stoichiometric equivalent between organic matter and carbon	0.1	0.1
O2LIM	DO concentration at which anaerobic processes begin, *gm*$^{-3}$	0.2	0.2

*Sets of 3 values represent values for the 3 algal groups (diatoms, greens, and cyanobacteria)
NOTE: The initial values were established based on the final values used for an area lake recently having undergone a WQ study, Center Hill Lake.

4.4.2.1 Biological Coefficients

The original Version 2.05 CE-QUAL-W2 model had only one algal compartment. Because algae are of particular interest in most reservoirs, Version 2.05 model was modified to have three algal compartments. This modification was patterned on the algal compartments used in Version 3.0 of CE-QUAL-W2. Data on the species composition of the algal community were available from 1981 through 1984 and 1996 through 1998. The three algal communities specified for the Dale Hollow Lake model are diatoms plus yellow or golden-brown algae (referred to as diatoms), green algae, and blue-green bacteria (cyanobacteria). The dominant phytoplankton species was determined by ranking the reported "count" of the samples collected at the dam (Station 3DAL20002) and on a midlake station (Station 3DAL20006), in each of the algal groups. Table 4.7 is a list of the dominant phytoplankton species from each of these communities observed in samples collected from Dale Hollow Lake.

Table 4.7. Dominant phytoplankton observed in Dale Hollow Lake grouped by taxonomic assemblage.

Diatoms	Green Algae	Blue-Green Bacteria
Cyclotella	Crucigenia	Cyanophyta
Centrales	Tetraedron	Anacystis
Dinobryon	Scenedesmus	Aphanocapsa
Pennales	Ankistrodesmus	
Synedra	Chlamydomonas	
Fragilaria	Eudorina	

Algal coefficient values are required for growth (gross production), mortality, excretion, respiration, and settling rates. Based on the algal populations sampled in Dale Hollow Lake, a median value was established for growth and respiration. The other coefficients listed above were based on published literature and previous water quality studies of similar water bodies. Parameter values are also required for the fraction of algal biomass that is converted to detritus, and for phosphorus, nitrogen, silica (diatoms only) and light half-saturation parameters. These parameters were set initially to values established for Center Hill Lake, which has similar properties. Changes to the values were made specific to the reaction of Dale Hollow Lake and are discussed in more detail in subsequent sections. Algal production in the model is not limited by carbon, so half-saturation values for this constituent are not required.

In general, values for algal related parameters and coefficients were obtained from Bowie et al. (1985), Cole (1995), Jorgensen (1979), Harris (1986), and Mills et al (1985).

4.4.2.2 Chemical Coefficients

Chemical rate coefficients include dissolved organic matter (DOM) and particulate (detritus) organic matter decay rates, SOD, and nitrification rates. DOM has two components: (1) a labile, easily metabolized fraction (LDOM), and (2) a refractory (RDOM), slowly metabolized fraction. Detritus also has an associated settling rate in addition to a decay rate. These rates were obtained from the CE-QUAL-W2 manual and literature cited above. Initial SOD rates were estimated based on experiences from previous water quality studies on similar water bodies. Based on calibration results, the SOD rates were set to 0.5, the median value of the suggested range. Nitrification rates represent the combined processes of nitrification (ammonia conversion to nitrite) and nitrafication (nitrite conversion to nitrate) because the model considers nitrate-N as one compartment.

4.4.3 Initial Conditions

The CE-QUAL-W2 model was used to model temperature, LDOM and RDOM, detritus, DO, ammonia-N, nitrate-N, phosphorus, iron, silica, and algae in Dale Hollow Lake. The in-pool concentrations of these constituents at the beginning of the simulation period (March 1) were defined in the model control file. Initial temperatures were discussed in Section 4.2.2. Initial concentrations for the remaining modeled parameters are discussed below.

The initial concentrations of DO, ammonia-N, nitrate-N, and phosphorus were set to the median in-lake surface measurements for the particular study year.

Initial diatom, green algae, and blue-green bacteria concentrations were based on the assumption of limited presence in the lake initially. To indicate their minor introduction to the lake via the inflow tributaries, the initial condition concentration of diatoms, green algae, and blue algae in the lake was set to 0.001 mg/L. It is generally understood that riverine algae will fall out quickly as the tributary enters the lake body, which is thereby indicative of the low algal source from tributary inflow. The initial concentration values used for silica, detritus, and iron were the computed median of the values used in the early part of the model year for all of the inflow tributaries. These calculations were made because there were no usable in-lake measurements on which to base the initial in-lake constituent concentrations for Dale Hollow Lake. The initial RDOM values were assumed to be 10% of the initial detritus, which was the same approach used in the development of the inflow concentration values. The LDOM was set to 0.1 mg/L based on previous studies. One exception to initial LDOM was during the 1988 study year, where the LDOM value was set to 0.05 mg/L. Typically LDOM will be less than RDOM, but in this study year the initial RDOM value equaled 0.05 mg/L. The initial LDOM value was then set to 0.05 mg/L. Table 4.8 includes a summary of the initial constituent concentrations used in Dale Hollow Lake per study year.

4.4.4 Water Quality Calibration Results

The questions being asked about Dale Hollow Lake water quality relate primarily to DO, nutrients, and chlorophyll *a* (as a surrogate for algae). Iron has also been included in this study. This section discusses the results from the calibration process for these constituents. Plots of predicted and measured concentrations of these parameters at the reservoir water quality monitoring stations are included in Appendix K.

4.4.4.1 Dissolved Oxygen

DO integrates many of the water quality processes occurring within the reservoir and provides more information about the conditions of a lake or reservoir than any other single constituent (Hutchinson 1957). The modeled DO profiles at the Dale Hollow dam include an additional monitoring station manned by the EPA during the 1973 study year. In all study years the measured and computed DO profiles at the dam stations were comparable. In the early spring of 1973, the model produced lower DO values than measured. However by mid-May the modeled profile was consistent with the measured data. During the late summer-fall of 1973, the modeled profiles agreed with the measured data as the hypoliminion approached anoxic conditions, and subsequent overturn. The 1973 study year represented wet conditions.

Table 4.8. Summary of initial in-lake constituent concentrations (CIC) per study year.

Dale Hollow Reservoir				
WQ Calibration; Initial In-Lake Constituent Concentration				
Constituent	1973	1988	1991	Comments
Tracer	0	0	0	
TSS	2	1	1.5	median value of inlake surface samples for study year
Coliforms	--	--	--	Not included in simulations
TDS	98	127	126	median value of inlake surface samples for study year
LDOM	0.1	0.05	0.1	based on RDOM values
RDOM	0.1	0.05	0.11	10% of detritus
Silica	1.2	1.5	1.5	based on the median of inflow msmt of DH tribs
Detritus	1	0.5	1.1	based on the median of inflow msmt of DH tribs
Diss Phos	0.01	0.01	0.01	median value of inlake surface samples for study year
Ammonia-N	0.1	0.2	0.2	revised based on model results
Nitrate+Nitrite	0.22	0.1	0.2	revised based on model results
DO	8.2	8.1	8.6	median value of inlake surface samples for study year
Sediment	--	--	--	Not included in simulations
TICarb	--	--	--	Not included in simulations
Alkali	--	--	--	Not included in simulations
PH	--	--	--	Not included in simulations
PCO2	--	--	--	Not included in simulations
HCO3	--	--	--	Not included in simulations
CO3	--	--	--	Not included in simulations
Iron	0.21	0.1	0.101	median value of inlake surface samples for study year
CBOD	--	--	--	Not included in simulations
Diatom	0.001	0.001	0.001	indicative of minor introduction from tributaries
Greens	0.001	0.001	0.001	indicative of minor introduction from tributaries
Blue-Greens	0.001	0.001	0.001	indicative of minor introduction from tributaries

The 1988 study year represented dry conditions and exhibited some unique results. For example, in mid-May the modeled profile indicated lower DO values throughout the profile. However by early July the measured and predicted profiles were comparable. In mid-September the hypolimnetic DO was just beginning to approach anoxic conditions, which was consistent with a dry year. In the wet year of 1973 and normal conditions of 1991 the hypolimnion neared anoxic conditions by late summer. During the normal year, 1991, the DO profiles at the dam were fairly consistent in shape. There was limited data for 1991 with only 2 measurements, one in late summer and one in late fall. The 1991 measured data for July indicated a metalimnetic increase which was likely attributable to an event.

The modeled DO profiles located at various lake stations indicated the relatively high DO values throughout the reservoir, as well as depicting the slight reduction of DO in the metalimnion, consistent with water quality reports published in 1976 and 1986. There were occasions when the computed profiles indicated the DO may be depleted in the metalimnion too early at some modeling stations, however at other stations the profiles were consistent with measured data. As indicated above, two separate water quality reports mentioned the high DO measurements of Dale Hollow Lake and the lack of oxygen depletion in the metalimnion. The District's 1975 report "Water Quality Conditions in Dale Hollow Lake" concluded the high DO values can be attributed to the physical processes of stratification. The report also indicated there probably exists a low level of organic sediments in the benthic zone. Typically, these characteristics are attributable to waters of high quality, which are low in nutrients and biological life. Modeled DO profiles for each of the study years supported the conclusions reached in these previous studies. The computed results indicated the model was reproducing the reservoir processes specific to Dale Hollow Lake. Plots of the modeled and measured DO profiles are included in Appendix K.

Figure 4.23 shows modeled release DO concentrations with measured DO concentrations downstream of Dale Hollow Dam. In the spring and early summer the model tended to predict release DO concentrations lower than those measured. This underprediction of tailwater DO concentrations was not surprising, even for a hydropower project since the model does not

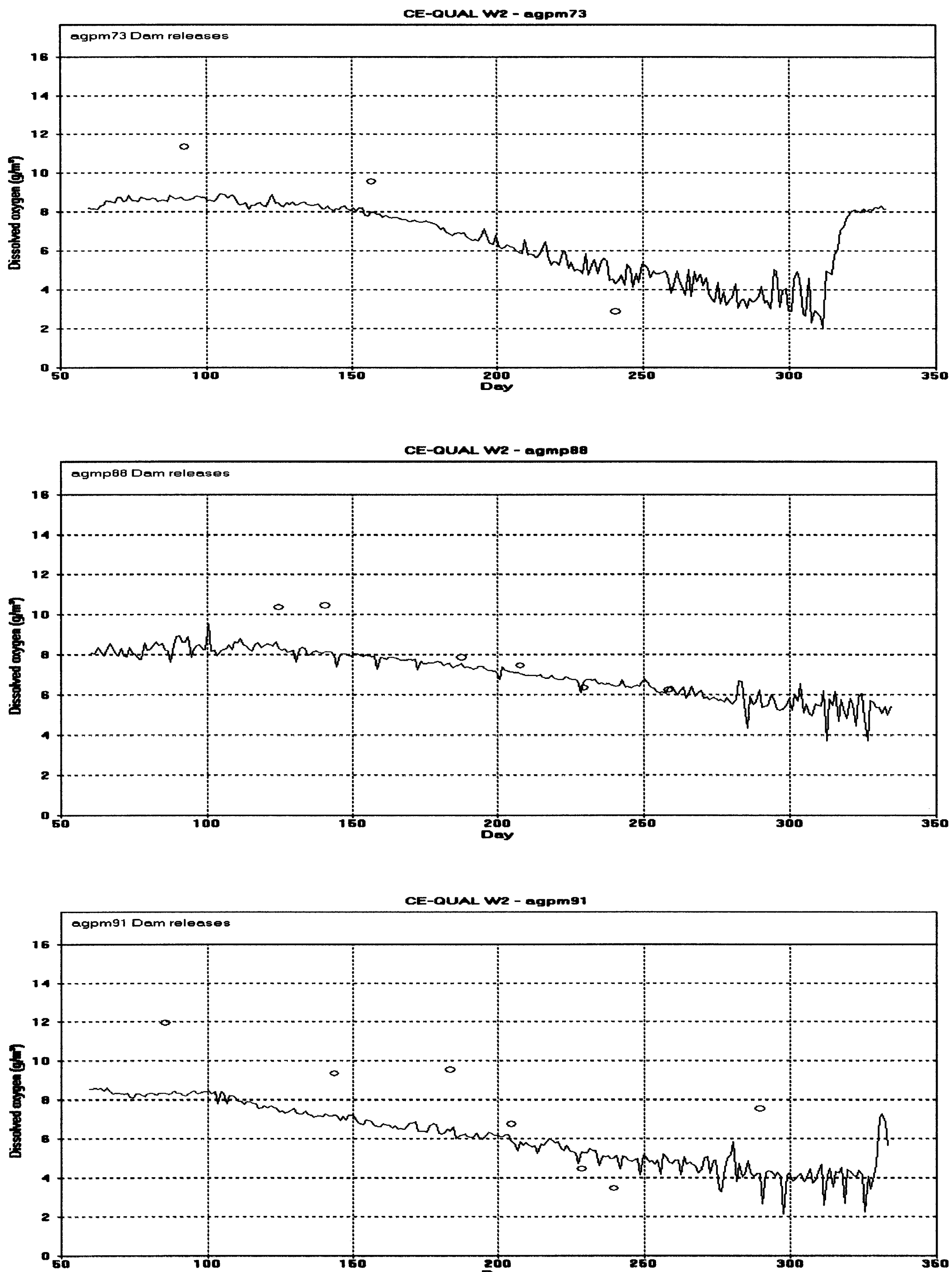


Figure 4.23. Comparison of measured and modeled DO downstream of Dale Hollow Dam.

assume any reaeration through the dam, nor between the dam and the downstream monitoring location. Predictions during mid to late summer are much better.

4.4.4.2 Dissolved Phosphorus

Measurements of orthophosphorus (1973) and dissolved phosphorus (1973, 1988, and 1991) in the reservoir were often less than detection. Phosphorus profiles indicated that phosphorus release from sediments did occur during extended periods of anoxia. Model phosphorus profiles were similar in shape to the measured profiles for the model years. The best dissolved phosphorus input data were from 1973, when inflow orthophosphorus was measured 3 times at the EPA monitoring stations and dissolved phosphorus was measured 3 times at the District monitoring stations. The model orthophosphorus profiles for 1973 showed responses to variability in the orthophosphorus load similar to those displayed in the measured profiles (orthophosphorus was measured only at the EPA stations during 1973). Dale Hollow Lake is a phosphorus limited lake, which is indicated in both the measured and computed profiles throughout the lake. The modeled phosphorus seemed to be low. However, with so much data reported at detection limit it was difficult to know if the modeled results were in fact correct. The phosphorus concentration does increase in the hypolimnion, coincident with sediment release under anoxic conditions for both measured and modeled results.

An unusual dissolved phosphorus condition existed on the Spring Creek tributary to Dale Hollow Lake. The orthophosphorus and dissolved phosphorus samples from Spring Creek yielded results 10 times greater than the measured data on the other monitored tributaries. The inflow concentrations of Spring Creek included these high values for all of the study years, so there was no basis for disregarding the measurements. Even with these high values, however phosphorus limitation was still prevalent. In the EPA's NES study of (EPA 1977) the high orthophosphorus levels on Spring Creek were attributed to a possible underestimation of the contribution of the Albany wastewater treatment facility or perhaps differing land-use practices in the drainage basin.

Plots of the modeled and measured dissolved phosphorus profiles are included in Appendix K.

4.4.4.3 Nitrogen

Reservoir ammonia-N concentrations reported were often at detection limit. The modeled ammonia-N values were typically lower than detection. There were several ammonia-N measured values that seemed to be high in comparison to the reported values at other monitored stations, which probably record an inflow event that the model was unable to duplicate. Ammonia-N profiles predicted that ammonia-N may be released from sediments when the hypolimnion is anoxic, which is consistent with the measured data. The model ammonia-N concentrations were typically low when compared to measured concentrations, but exhibited increases in hypolimnion concentrations due to sediment releases similar to the measured data. Numerous attempts were made to increase the modeled ammonia-N while maintaining reasonable coefficients. These attempts included raising inflow concentrations and initial concentrations of ammonia-N. The computed ammonia-N values did not increase as a result of these changes. Generally the concentration of ammonia-N is low in well-oxygenated or unproductive oligotrophic waters, which is consistent with Dale Hollow Lake (Wetzel 1983). In addition, ammonia-N is rapidly assimilated by algae and represents the most significant source of nitrogen for plankton in many lakes (Liao and Lean 1978).

Measured reservoir nitrate-N concentrations tended to be pretty erratic throughout the study years. The 1973 measured data offered the greatest number of sampling stations where modeled and measured data matched relatively well. A storm event in June 1973 produced high concentrations that the model was not able to reproduce. During 1988 and 1991 the modeled data seemed to mimic the shape of the measured data although tending toward the high side of concentrations. In general, it seemed during the three study years there was a tendency for the modeled data to be higher than the measured data in the epilimnion but match the measured data in the metalimnion and hypolimnion.

The NES report, based on the water quality sampling of 1973, indicated that there was a loss of nitrogen during the year. The 1991 samples seem to support the loss of nitrogen through the year, however because there was such limited data it was difficult to fully support their conclusion. In 1988 there was not enough data to support or refute the loss of nitrogen throughout the year.

Plots of the modeled and measured ammonia-N profiles and nitrate-N profiles are included in Appendix K.

4.4.4.4 Chlorophyll *a* (algae)

Algae are represented by three model compartments: 1) diatoms, 2) green algae, and 3) blue-green bacteria. While the model maintains a mass balance on algal carbon, the typical field measurement of algae is one of the algal pigments, usually chlorophyll *a*. Chlorophyll *a* measurements were not taken by the District in the reservoir during 1973, but they were measured by the EPA for the NES study during 1973. Chlorophyll *a* was measured by the District and KY in the reservoir during 1988 and by the District only during 1991. The model algal carbon concentrations were converted to chlorophyll *a* by dividing the concentrations by 65 (Cole 1995). The sum of the estimated chlorophyll *a* concentrations for the three algal groups was compared to measured chlorophyll *a* concentrations. Because of spatial patchiness, chlorophyll *a* had a relatively high variance in field samples, averaging around $\pm 15\ \mu g/L$ (APHA 1995). Plots of modeled and measured chlorophyll *a* profiles are included in Appendix K.

There were very little data to represent measured chlorophyll *a* profiles. During 1973 the District did not measure chlorophyll *a* and the EPA in-lake sample results were integrated into a single reported value. The maximum depths sampled were 34.1 meters at Station 210201, 36.0 meters at Station 210202, 29.9 meters at Station 210203, 17.1 meters at Station 210204, 21.6 meters at Station 210205, and 12.2 meters at Station 210206 (EPA 1977). The 1973 measured data therefore prevented a direct profile comparison of modeled and measured chlorophyll *a*. In order to compare 1973 measured and modeled data the time series plots were utilized. Time series plots represent the mean modeled chlorophyll *a* values from the top 2 layers (i.e., 2 m) of the lake at stations representing the upstream reach of the lake, the downstream area and the mid-lake region. The measured data from the top 2 layers was also represented at the station representing these regions of the lake. In all of the study years the average variance of measured to modeled chlorophyll *a* was less than 5 $\mu g/L$, which is less than two-thirds the expected variance from field sampling.

During 1973 on the upper reach of the lake the expected chlorophyll *a* concentration is approximately 8 μg/L greater than measured. This maximum difference appears to be a timing issue, as modeled concentration drops off sharply relatively quickly to a value that actually matches the measured data.

An additional graphical anomaly exists at the lower lake region during 1973. The graphical representation of chlorophyll *a* concentrations indicates a drop from nearly 23 μg/L to 0 μg/L in early April. By reviewing the animation of the model output this rapid drop in chlorophyll *a* concentration is not really correct, but rather due to wind and mixing that has created a reverse movement in the upper layers and thus pushed the algal group upstream of the dam so that it is not reflected in the segment being plotted. The actual chlorophyll *a* concentration has not dropped out, just dropped back and out of the plot window (Chlorophyll *a* time series plots are included as Appendix L).

When there were available profile measurements for 1988 and 1991 the modeled chlorophyll *a* profiles tended to exhibit lower values than the measured values in the epilimnion, but comparable to the observed data in the metalimnion. In all cases, the modeled chlorophyll *a* concentrations were near the concentration observed. However, the elevation of the maximum growth was lower than observed. The 1988 and 1991 profiles indicated that the modeled chlorophyll *a* maximum growth concentration tended to be 5-15 ft below the measured growth maximum elevation. One possible reason for the lower elevation growth maximum in the modeled data would be due to light inhibition in the upper layers, thereby pushing the algae deeper, where the lower light proved optimum for greater growth. A sensitivity analysis was performed on the coefficient of light saturation. The results of the sensitivity run indicated that, while the light saturation coefficient affected the maximum concentration, the elevation of the maximum growth was not affected. An alternative sensitivity run was then done on the temperature ranges of the curve defining the effects of temperature on algal growth rates. A wide range of temperatures did not alter the elevation of the maximum growth, it only inhibited or stimulated growth. The coefficient of light saturation and the temperature range defining the algal growth rate curve were then reestablished to within reasonable range where the concentration of algal growth was best represented in the measured data.

Time series plots of the predicted concentrations for the three algal groups were also informative (Appendix M). These plots showed the relative abundance and dominance succession for the three algal groups in the top 2 layers of the lake. The accepted model of algal succession proceeds with diatoms dominating in the winter and early spring when water temperatures are cool, followed by green algae in the late spring and early summer, then blue-green bacteria in the late summer, and diatoms becoming dominant again in the fall with cooler water temperatures. The model results at the dam generally follow this pattern, with the exception of the blue-green bacteria in the late summer, which was not predicted. The likely reason for the absence or small amount of blue-green bacteria was the phosphorus limitation of Dale Hollow Lake. By the late summer the diatoms and green algae have used up the supply of phosphorus making the emergence of blue-greens more difficult. During 1991 in the upper reach of the lake, there was a larger concentration of green algae than diatoms early in the spring. This concentration pattern corresponded to a rather large inflow event early in the spring of 1991. Algal identification and enumeration has been conducted at the reservoir water quality monitoring stations. However, the enumeration results are not consistently available for all depths, so it is difficult to determine if algal succession in Dale Hollow Lake follows the accepted model.

The maximum modeled chlorophyll *a* concentration output during the 3 study years occurred in the upper portion of the lake in 1991 (30 μg/L) and was similar to the reported historical maximum of 32 μg/L.

4.4.4.5 Iron

Total Iron was included as a water quality constituent to provide the capability to simulate the buildup of dissolved iron and manganese in the hypolimnion due to anoxic conditions and the release of these constituents. Reservoir iron concentrations were measured irregularly during the study years and often were equal to or less than the minimum detection limit. Given the limited data, the model output adequately represents the total iron profiles. Increased concentrations occurred throughout the year in the anoxic hypolimnion in the upper reservoir reaches. These predicted results were corroborated by representative measured data.

4.4.5 Conclusions

The calibration exercise indicated that the model predicted the general patterns and seasonal changes expected in reservoir water quality, generally agreed with measured water quality, and represented the appropriate limnological processes. The calibrated Dale Hollow Lake model appeared to reproduce processes affecting DO and nutrient concentrations well. Calibration of algae was difficult. Although the algae calibration results were not as good as the other constituents, model chlorophyll *a* concentrations are generally within the range of analysis variability from measured concentrations. This calibrated model is a viable tool for evaluating different reservoir and watershed management alternatives.

5.0 SCENARIO SIMULATION

The calibrated model was used to evaluate two scenarios on Dale Hollow Lake. The first scenario evaluated the impacts of existing major point sources of pollution on the water quality of Dale Hollow Lake. The second scenario evaluated the impacts from houseboats discharging sanitary wastewater on the water quality of Dale Hollow Lake. Both scenarios were simulated for the dry year (1988).

5.1 Scenario One: Major Point Source Removal

Scenario One involved removing the inputs to the reservoir from major point source dischargers on Dale Hollow Lake and its tributaries. These major point source dischargers include three wastewater treatment plants (WWTP) in the watershed. The NPDES discharge list included in Appendix A includes the WWTP facilities at Albany, KY, Byrdstown, TN, and Jamestown, TN.

5.1.1 Scenario One Inputs

The tributary water quality monitoring stations used to develop input water quality for the Dale Hollow Lake model were located downstream of the WWTP's discharging in the watershed (refer to Figure 2.1). Therefore, in the calibrated model, the water quality inputs include the contributions from these point source dischargers. The distance between the dischargers and Dale Hollow Lake was generally significant, however a conservative approach was applied by inputting the nutrient concentrations without consideration of assimilation or uptake from their point of discharge to the tributary waters where they are being input in the lake. As mentioned above, three WWTPs used in this scenario are located in Jamestown and Byrdstown, TN and Albany, KY. The ultimate receiving streams from these WWTPs that flow into Dale Hollow Lake are the East Fork Obey River, Wolf River, and Spring Creek respectively.

The annual loading for the respective tributaries is the product of average daily flow and daily concentration values used as the input values for the 1988 calibrated model, and then converting to an average annual load. During 1988 discharge monitoring reports (DMR) were not available as a source of water quality data from which to calculate annual nutrient loads originating from the WWTPs. For ammonia-N, the annual nutrient load contributed by the WWTPs was calculated using the permit limit designated on the most current DMR. For phosphorus loading, it was assumed the WWTPs operated such that a 25% removal rate was accomplished, assuming conventional treatment plus a trickling filter (Metcalf & Eddy 1991). The nitrite-N loading from untreated wastewater was zero, per Metcalf & Eddy (1991) and was therefore not included in further computations. The annual loading contribution from the WWTPs was computed and the inflow concentration values were lowered the appropriate percentage to reflect the absence of these major point sources in their respective receiving streams. The reduction applied only to phosphorus and ammonia-N, nitrite-N was left unchanged as indicated above and no other parameter was modified. A summary of the computed average annual loading and the subsequent load added by each of the treatment facilities is included in Table 5.1.

Table 5.1. Annual nutrient loads for Dale Hollow Lake.

WWTP FACILITY	RECEIVING STREAM	WWTP FLOW (mgd)	PHOSPHORUS (1988 Stream Load/ WWTP annual load[1], g)	AMMONIA-N (1988 Stream Load/ WWTP annual load[2], g)	NO_2+NO_3 (1988 Stream Load/WWTP annual load[3], g)
Jamestown, TN	East Fork Obey River	0.63	0.085/0.062 WWTP=73%	0.845/0.052 WWTP=6%	3.14/0 WWTP=0%
Albany, KY	Spring Creek	0.45	0.052/0.044 WWTP=85%	0.109/0.079 WWTP=72%	1.36/0 WWTP=0%
Byrdstown, TN	Wolf River	0.25	0.029/0.025 WWTP=84%	0.353/0.022 WWTP=6%	1.35/0 WWTP=0%

[1]Phosphorus removal based on conventional treatment+trickling filter (15%+10% removal) (Metcalf & Eddy 1991)
[2]Limits taken from permit
[3]Based on concentration in untreated domestic wastewater (Metcalf & Eddy 1991)

5.1.2 Scenario One Results

The results of the major point source discharge nutrient load reduction are not reflected at the dam, however impacts are visible in the upper lake areas. The effect of removing point source loads was evaluated by comparing simulation profiles and time series plots for chlorophyll *a*, algae, phosphorus, ammonia, and DO to the same profiles from the 1988 dry year calibrated model. These profiles/plots are included as Appendix N. The comparison of the time series plots of chlorophyll *a* depict concentration reductions in the upper layers of the lake in the upper- and mid-lake stations resulting from the removal of the major point source loading. The reduction realized in the upper lake is nearly 20% and drops to 4% at the mid-lake station, 3DAL20004. The lower lake chlorophyll *a* values are not impacted in the spring, however a slight increase is shown during what would be the fall algal bloom. The comparison of the profile output of chlorophyll *a* indicates the decrease in chlorophyll *a* concentrations at many of the monitoring stations located throughout the lake. The effects of the point source discharge removal ranges from 75% to 40% reductions in peak concentrations near the mouth of the contributing tributaries. These reductions are evident as far downstream as monitoring station 3DAL20004, where a 12% reduction is reflected in the summer profile. Chlorophyll *a* concentrations at the dam remain unchanged. The profiles also show that the elevation range of maximum production is slightly increased. This can be attributed to an increased region of favorable growth, i.e., light. Algal group composition was unchanged but the total concentrations were reduced by nearly 50% in the upper lake area. By the mid-lake the reduction was slight. No effects were visible at the dam. The phosphorus profiles projected with the removal of the point source dischargers indicate limited changes from the baseline model, and then only in the areas of the impacted tributaries. The same is true with ammonia-N concentrations, which are even less impacted by the removal of the WWTPs. The DO profiles indicate localized changes. The model predicated a slight increase in the amount of DO in the metalimnion at sampling stations near Wolf River in the spring/summer. No other changes in the DO profiles were significant.

A plot of secchi depths predicted at the upper lake station with and without point source loading was developed as Figure 5.1 to provide additional data. The plot shows that lake clarity is improved from spring until late summer when point source loading is removed. The most improved clarity of 0.2 meters (approx. 8 inches) occurs in early May. Secchi depth is not part of the standard output from the model. The predicted secchi depth values were obtained from the model output by incorporating an additional command into the CE-QUAL-W2 program code. This model revision is included in the CE-QUAL-W2 revision documentation in Appendix B.

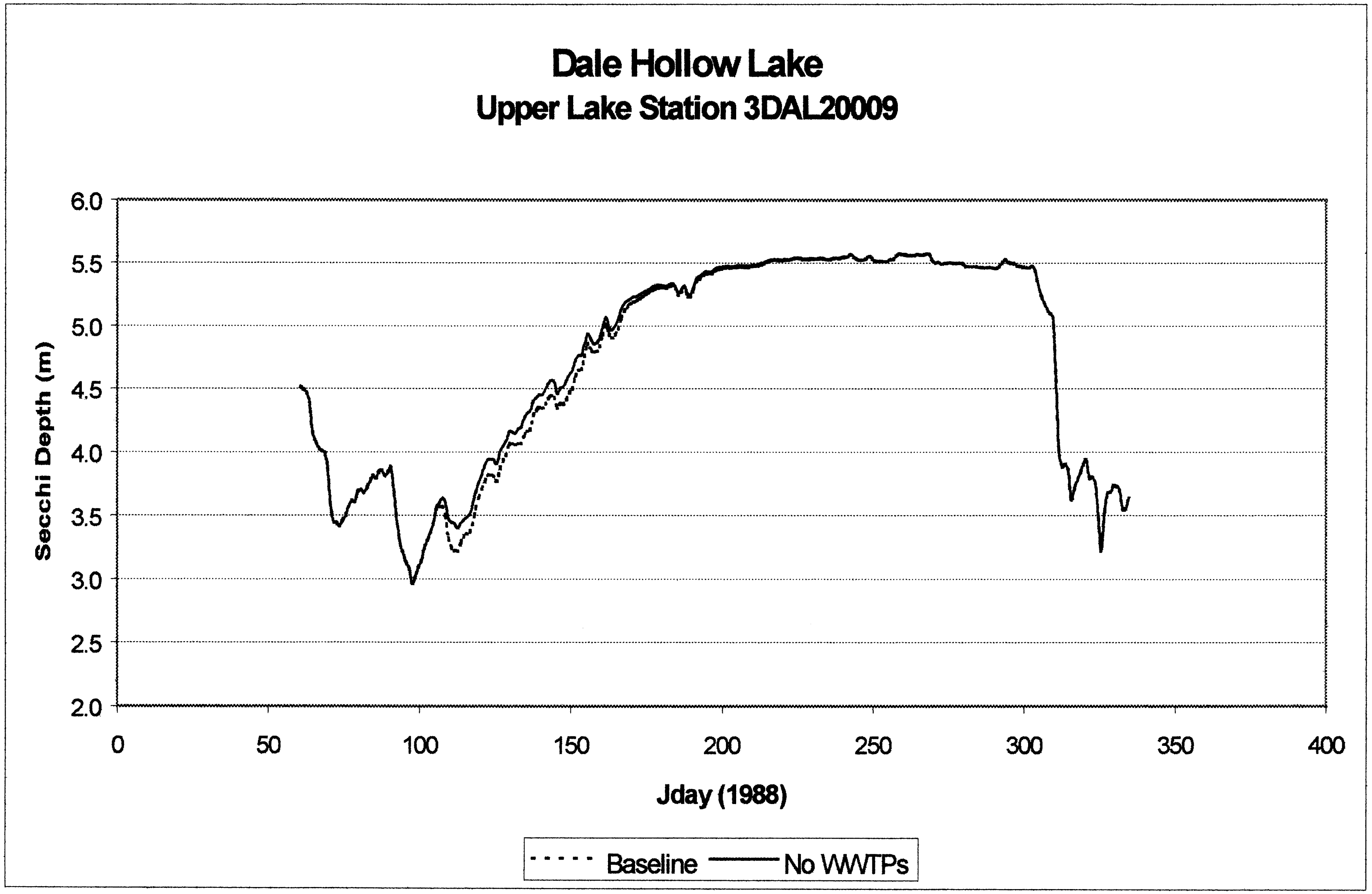


Figure 5.1. Secchi Depth on upper Dale Hollow Lake baseline vs. no WWTPs.

In summary, removal of major point source dischargers from the Dale Hollow watershed do impact the lake on its upper reaches by reducing the peak nutrient concentrations. This reduction in nutrient concentrations is reflected in total algal concentration and chlorophyll *a* concentrations. Point source reductions are most important during the growing season when the nonpoint source loading and storm loadings are less. The greatest impact is reflected in the upper part of the lake because the stream moves as an interflow below the plunge point. The effect of the point source removal is no longer visible at the dam station or below the dam.

5.2 Scenario Two: Houseboat Effects

Scenario Two investigated water quality impacts resulting from sewage discharge from houseboats. A number of houseboats were "placed" on two representative coves of the Dale Hollow Lake. The Holly Creek embayment and Sulphur Creek embayment represent a small and medium/large cove respectively. The houseboats are considered a point source discharger due to dumping of sanitary wastewater directly in the lake. Although sanitation dumping stations are located on Dale Hollow Lake, the dumping is known to occur.

5.2.1 Scenario Two Inputs

By adding a number of houseboats to a representative cove on Dale Hollow Lake, the water quality impacts of dumping was evaluated. This scenario was evaluated by assuming that one houseboat contains 12 persons and is located on a particular cove Friday through Sunday every weekend from May to October. Additional days are added for Memorial Day, July 4^{th}, and Labor Day holidays. The flow generated from each houseboat was assumed to be 40 gallons per day per person (Metcalf & Eddy 1991). The nutrient concentrations added as a point source were 3 mg/L of phosphorus and 25 mg/L of ammonia-N. Nitrite-N is not present in a measurable concentration in raw domestic wastewater (Metcalf & Eddy 1991). Nutrient concentrations were incorporated with the additional flow into new input files. These houseboat inputs were modeled as distributed tributaries to the embayments.

According to lake managers, during popular holiday weekends, such as Labor Day, approximately 1,000 houseboats might be anchored on the lake. Using this estimate, the average area per houseboat was calculated for the lake. The surface area of Dale Hollow Lake is approximately 30,990 acres. If 1,000 houseboats occupy the lake, a houseboat will be found every 31 acres. The approximate areas of Holly Creek and Sulphur Creek are 139 and 691 acres respectively. Using these values the estimated houseboat populations during heavy usage is 4 houseboats on Holly Creek and 22 houseboats on Sulphur Creek.

Three simulations were carried out using the 1988 calibrated model. The first and second simulations applied to the small cove, Holly Creek, and included 5 and 10 houseboats respectively. A third simulation was applied to the larger cove, Sulphur Creek and included 25 houseboats. The actual number of houseboats in a specific cove may actually be much larger than simulated, and any discharge from the houseboats may be more concentrated than indicated in the model simulation.

5.2.2 Scenario Two Results

The results of the increased nutrient load from a large population of houseboats on either a small or medium-large cove were not visible when compared to the 1988 calibrated model. A direct comparison was made of chlorophyll *a*, ammonia-N, and phosphorus concentrations in each of the respective embayments with the 1988 results at a depth of 1 meter. The predicted nutrient concentrations remained unchanged or exhibited minimal change in concentration. DO was also reviewed and the results were similar to that of the nutrients discussed above (i.e., no visible change). The time series comparison plots are included as Figures 5.2-5.8.

It appears from this scenario comparison that the current water quality is not visibly degraded with the current population of houseboats. This particular scenario set up was based on the approach that each of the houseboats present on the representative coves were discharging untreated domestic wastewater directly into the lake on a regular basis. Water quality degradation from houseboats is not immediately visible in the simulation results.

An animation of the embayments, by means of the AGPM program, was also reviewed comparing the calibrated "baseline" model and each of the three houseboat applications. The animations indicated a visible impact was not apparent with the inclusion of the houseboats sewage discharge. Profile Plots for the Fourth of July and Labor Day holiday weekends are included in Appendix O. Again, no changes are visible in the model output.

The model does include a limitation with regard to this scenario. The inflows containing the increased loads are instantaneously mixed in the waters of the receiving cells, thereby allowing a great deal of dilution to occur. A point source waste discharge from a houseboat is highly concentrated constituents being introduced into a small area. Some of the constituents present in this inflow may in fact be causing water quality problems in localized areas that are too small for CE-QUAL-W2 to accurately model.

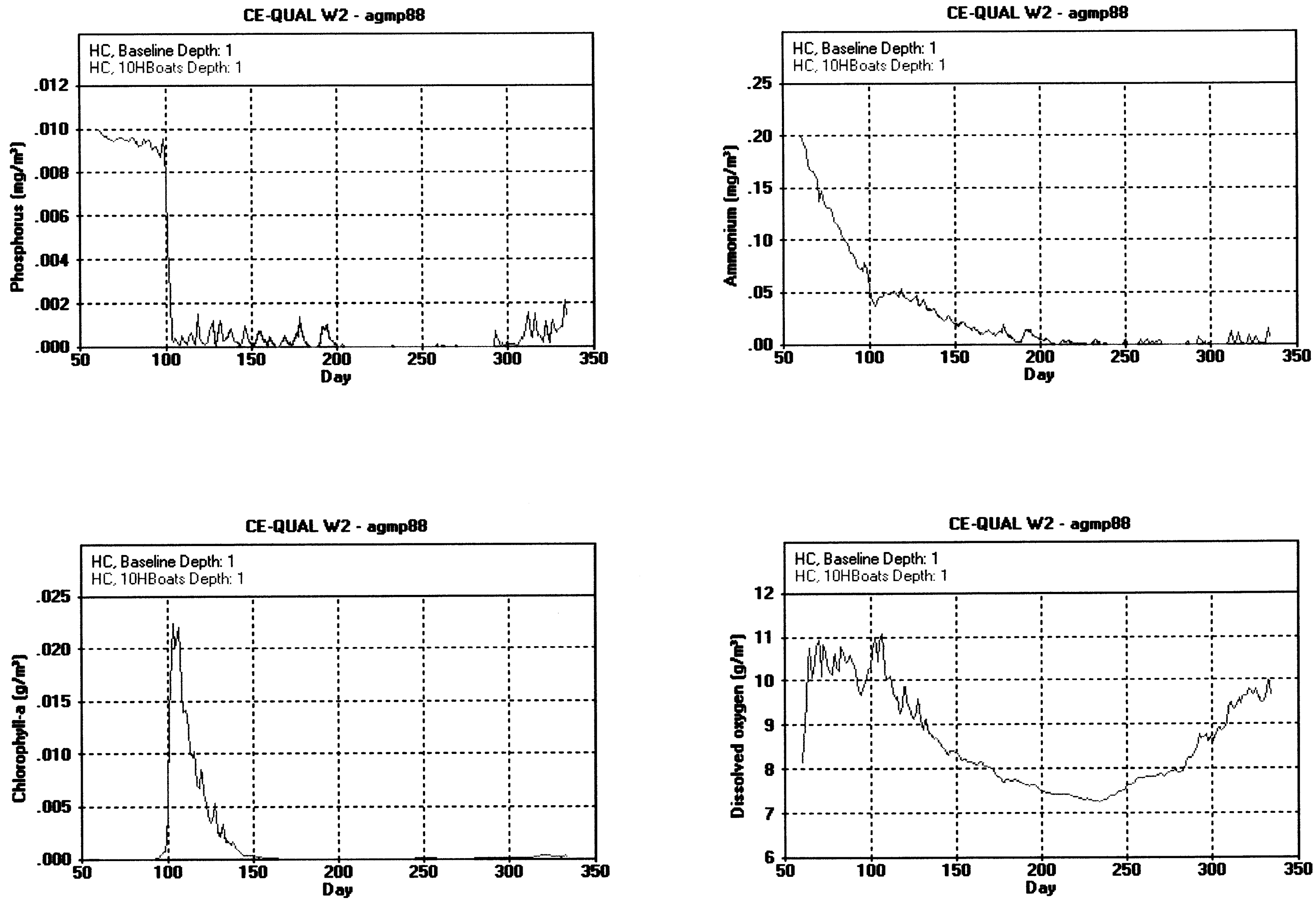


Figure 5.2. Holly Creek, lower segment, baseline vs. 5 houseboats.

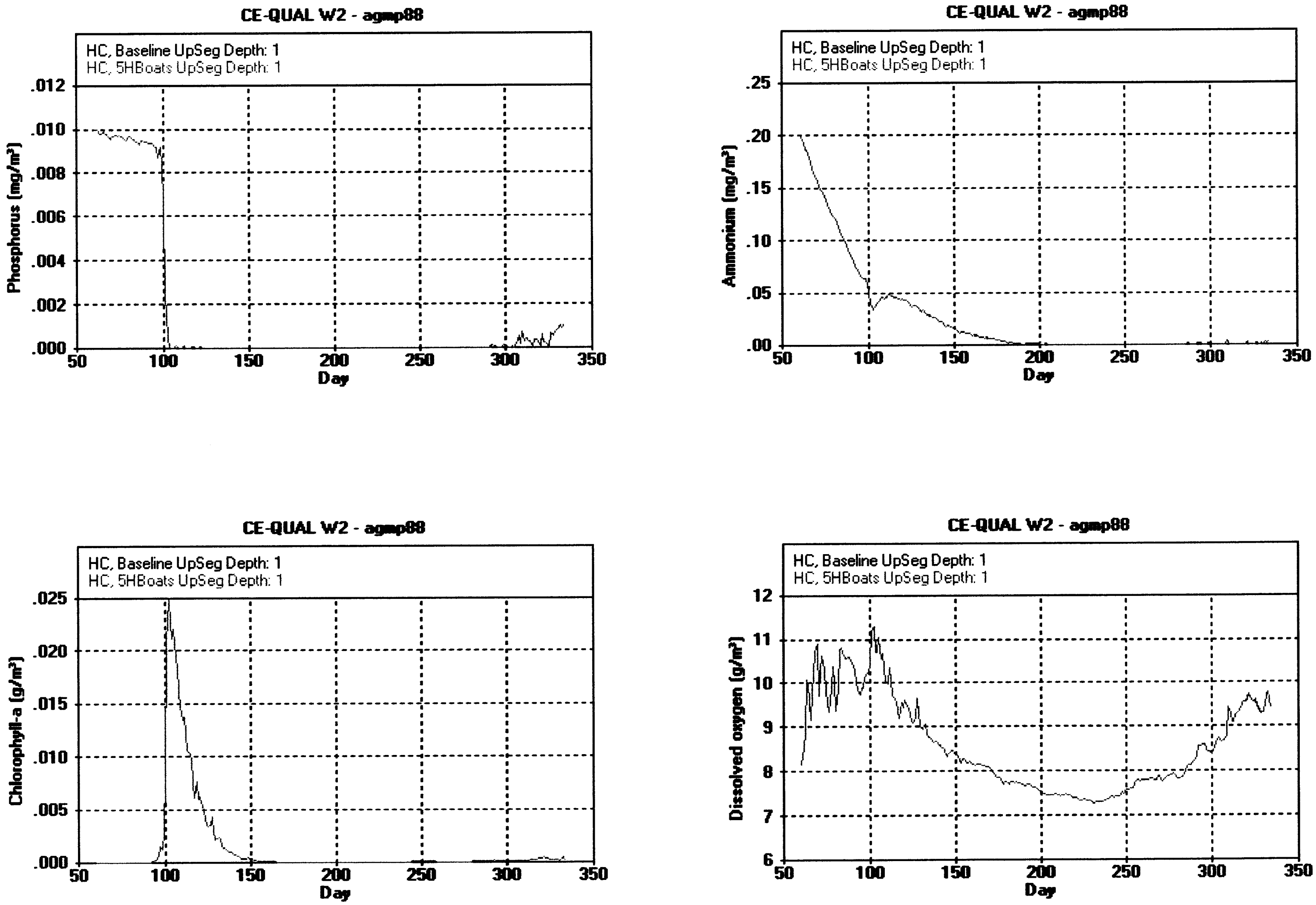


Figure 5.3. Holly Creek, upper segment, baseline vs. 5 houseboats.

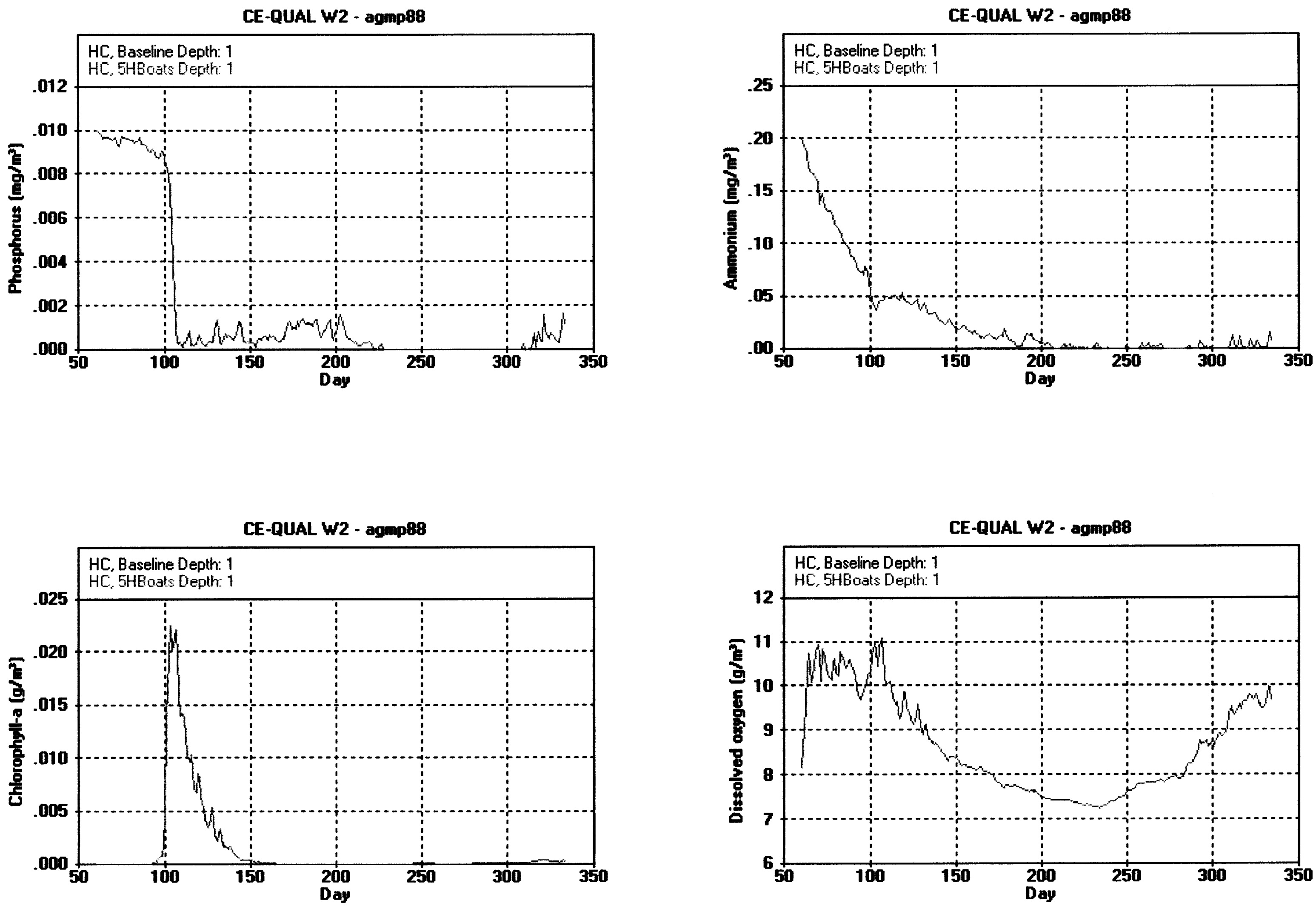


Figure 5.4. Holly Creek, lower segment, baseline vs. 10 houseboats.

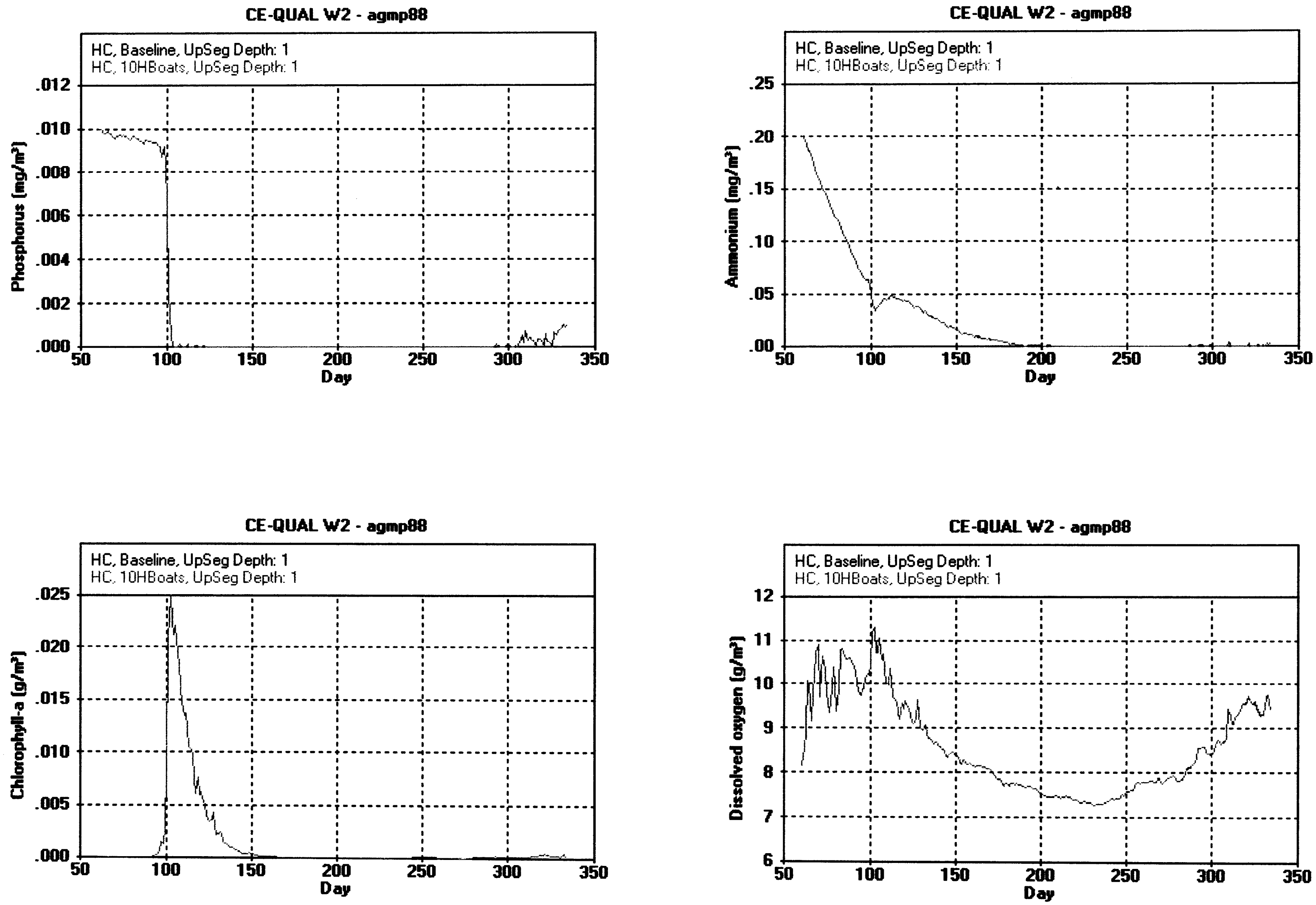


Figure 5.5. Holly Creek, upper segment, baseline vs. 10 houseboats.

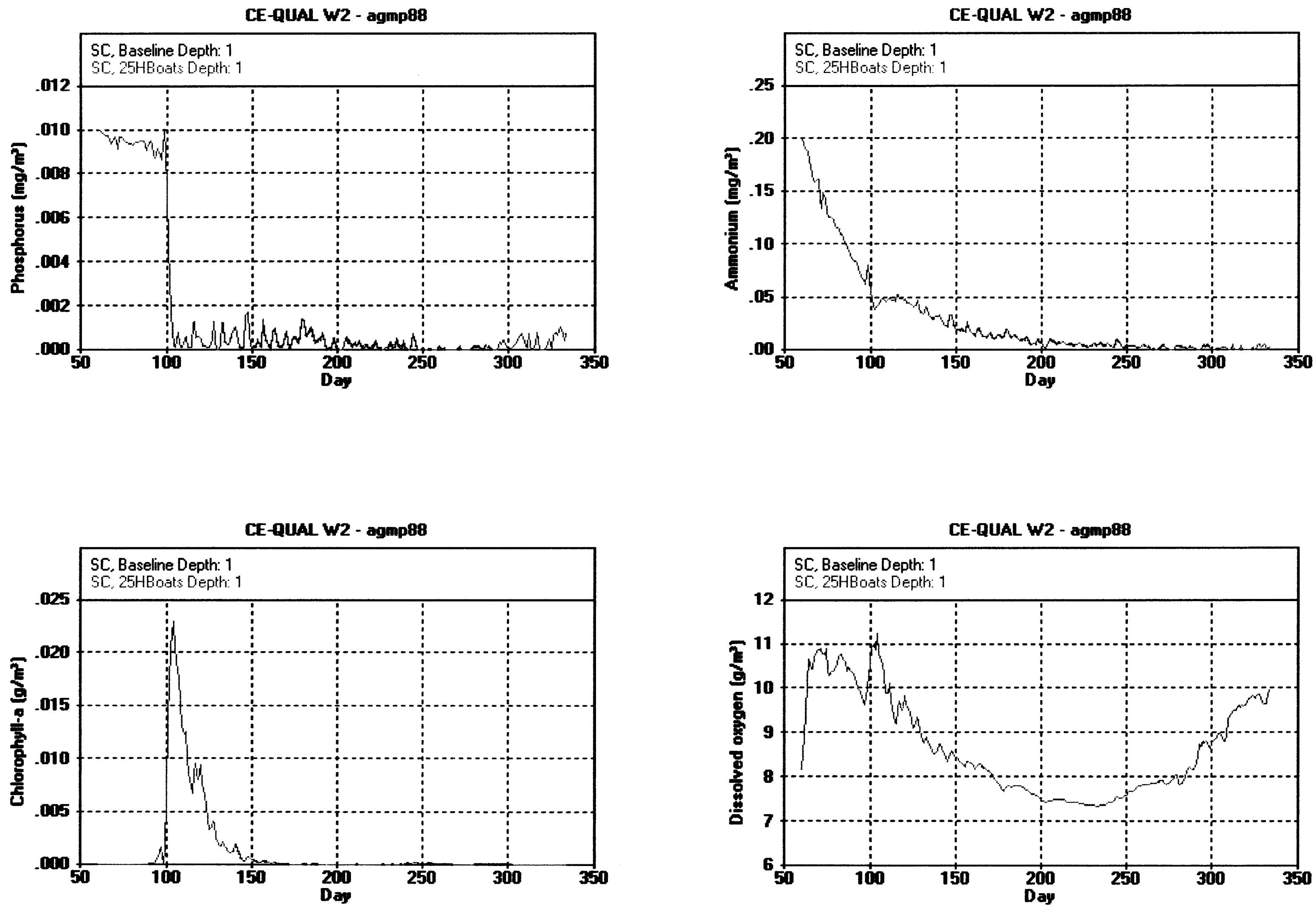


Figure 5.6. Sulphur Creek, lower segment, baseline vs. 25 houseboats.

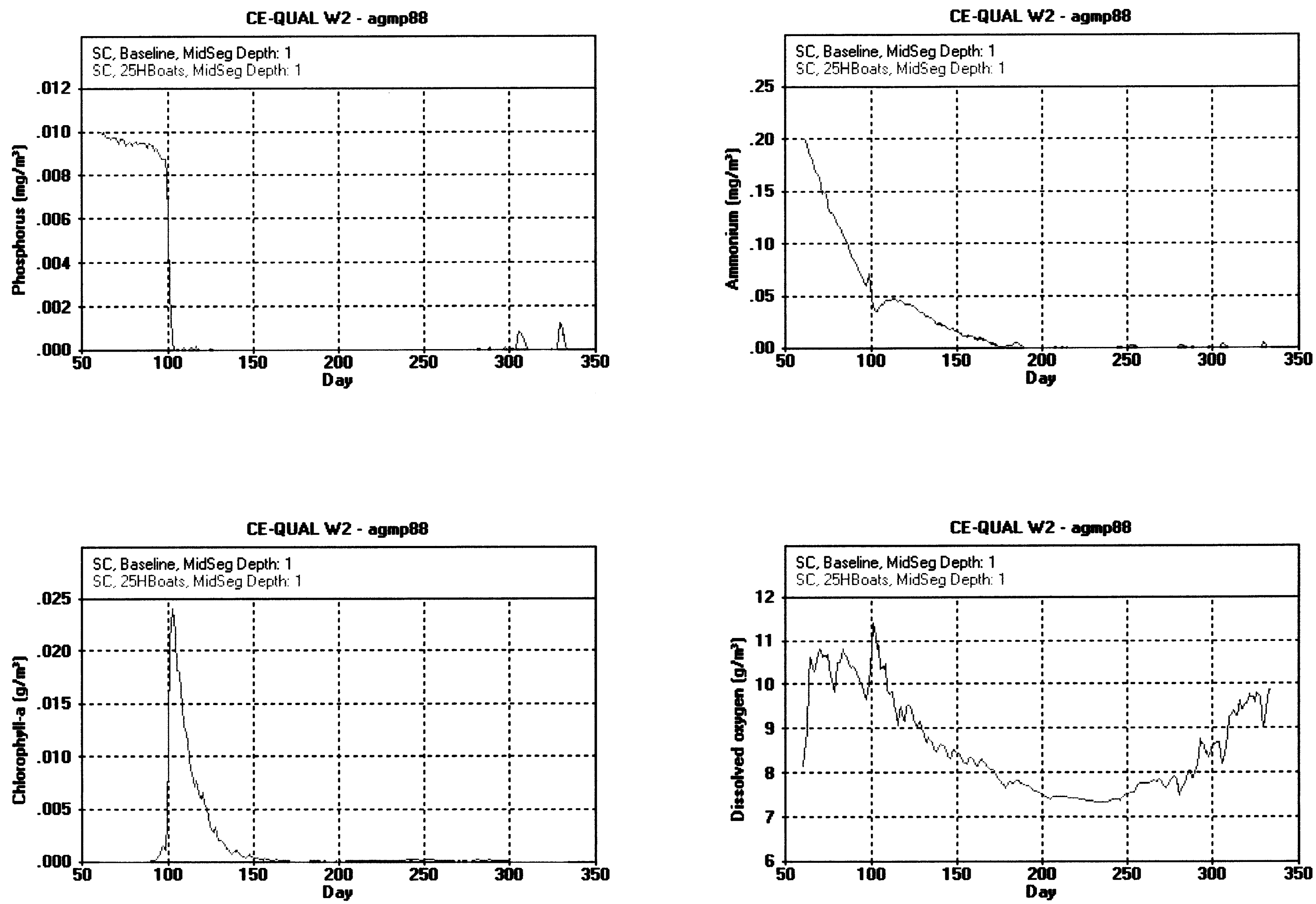


Figure 5.7. Sulphur Creek, middle segment, baseline vs. 25 houseboats.

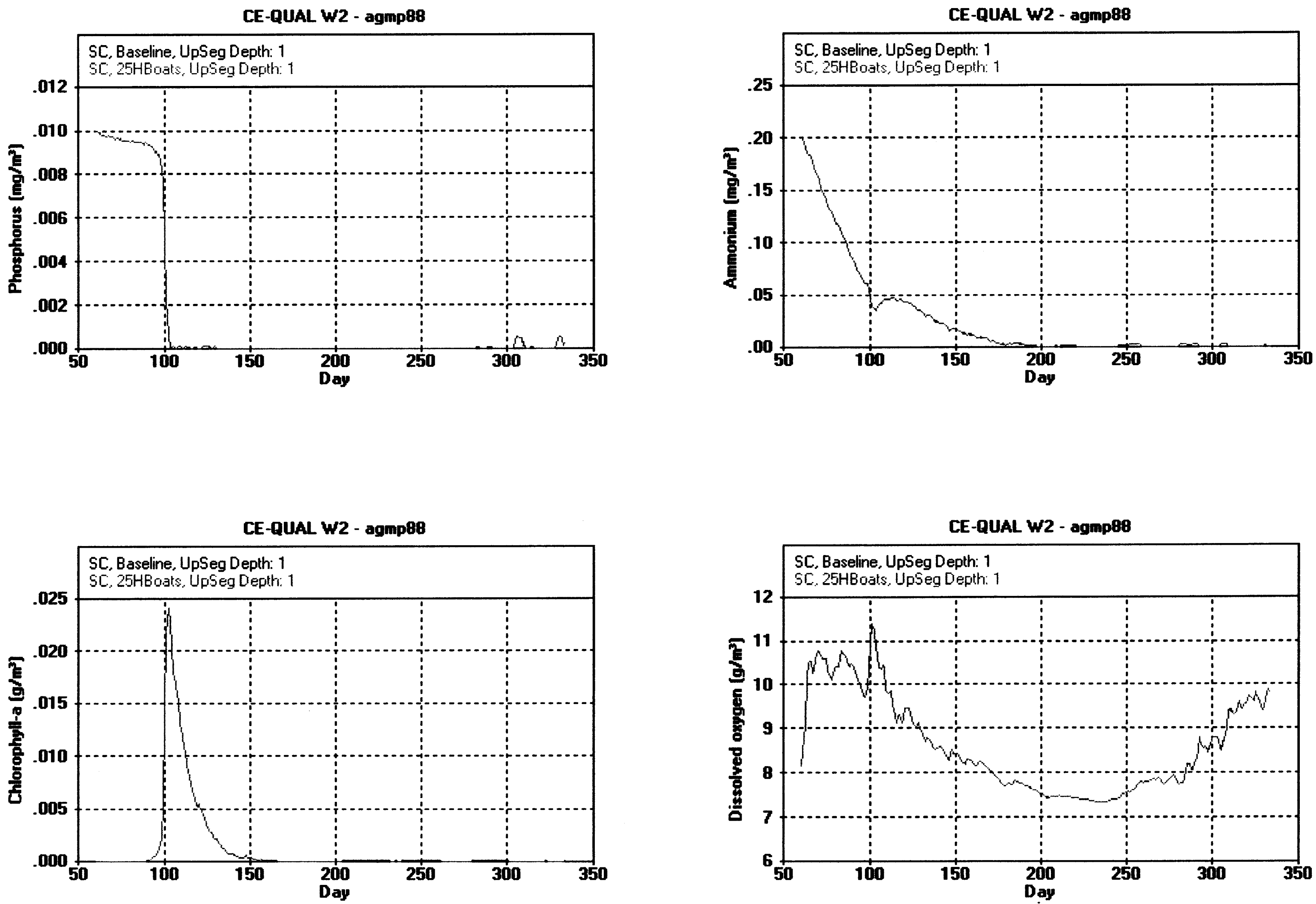


Figure 5.8. Sulphur Creek, upper segment, baseline vs. 25 houseboats.

5.3 Summary of Scenario Results

Two scenarios were applied to the Dale Hollow Lake calibrated "dry year" model to evaluate their impacts. Scenario One included the removal of identified major point source dischargers into the streams of Dale Hollow Lake. The results indicate that significant changes in the concentration of nutrients are reflected in the upper reaches of the lake. The effects became immeasurable at the dam. Water clarity appears to be reduced in the spring on the upper reaches of the lake, but no change is visible throughout the remainder of the year. It seems evident that concerns for decreased clarity in the upper reaches may be related to point source discharge.

The second scenario applied to houseboats on Dale Hollow Lake and their discharging of sanitary wastewater into two coves of the lake. The Holly Creek and Sulphur Creek embayments represent a small and medium/large size cove, respectively. A specific population of houseboats was determined and their representative load calculated and added to the baseline model. The results indicate that the simulated population of houseboats did not visibly impact the water quality.

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